ORGANIC MATTER DYNAMICS IN WILLOW AND EUCALYPT LINED CENTRAL VICTORIAN STREAMS

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Abstract

The spread of exotic willows (*Salix* spp.) in SE Australia, New Zealand and South Africa has provoked widespread debate in scientific, management and broader communities. In Australia, the extent of spread is unknown, but at least 30000 km of river frontage in Victoria are lined by willows. Management and research literature has identified the poor knowledge of willow impacts on Southern Hemisphere aquatic ecosystems. It has been speculated that the major distinction between deciduous willows and evergreen native vegetation will be the timing and quality of litterfall. This would have flow-on consequences for metabolic processes, stream biota and water quality at reach, stream and catchment scales. These two vegetation types were studied through the preparation of partial organic matter budgets for native and willow lined reaches in a central Victorian catchment. Organic matter inputs from litter, groundwater and gross primary production (GPP), organic matter standing crop and respiratory output were quantified.

Total inputs to willow and native reaches were similar (735 and 764 g ash free dry weight m^{-2} y^{-1}, respectively). Inputs were dominated by litterfall (~60%) and there were no significant differences in annual litterfall between sites. GPP contributed ~20% of total inputs and estimates suggested there were few significant differences in annual GPP, 24 h community respiration, ratio of GPP to community respiration (P/R) or net daily metabolism (NDM) between sites. Groundwater contributed ~20% of total inputs with one third of the dissolved organic matter sourced during short flow paths through riparian sediments. Aggradation at willow sites appeared to increase the riparian flow path. Willow and native sites were heterotrophic and similarly dependent on allochthonous organic matter (P/R=0.2, NDM= -1.6, and ratio of net primary production to total inputs ~0.1).

Estimates of metabolic processes at willow and native sites were very different to those at cleared sites. Clearing of ~10% of the catchment's streams had a measurable influence on indices of stream condition that are dependent on litter accession and community metabolism. The effects of willow clearing on reach and catchment metabolic function needs further research.

Standing crop of organic matter was the key point of distinction between willow and native sites. Benthic organic matter (BOM) was at least ten times greater at willow sites. Willow root mats, large woody debris (LWD) and debris dams exerted geomorphic control over step-pool sequences and contributed to high retention capacity and channel aggradation. Native sites lacked substantial substrate aggradation and were bedrock controlled. Material cycling indices based on BOM (specific respiration and BOM turnover) were very different between native and willow sites yet stream metabolism index was similar. The apparent anomaly was explained by at least one unquantified organic matter input (willow root mat) and, possibly, trapping by willow root mats of seston exported from less retentive reaches.

Greatest litterfall at both native and willow sites occurred in summer and autumn during times of low flow. The accumulation of litter in and alongside stream channels during this period led to a pulse of litter as flow returned in autumn. This pattern is typical of small and temporary streams flowing through native forests in south eastern Australia. Consequently it is predicted there will be few biotic consequences directly attributable to the timing of litterfall from willow riparian forests in such streams. The guiding postulate that timing of litterfall would be a distinguishing feature was questioned. However, the quality of litter, as indicated by ash content, was significantly different at willow sites. Biogeochemical studies focussing on spiralling of elements known to limit in-stream biological processes (eg nitrogen and phosphorus) are recommended.

Willow research and management should also focus on retention capacity, including the recruitment and role of LWD and the structure and function of root systems. Removal of willows potentially facilitates native vegetation establishment but simultaneously decreases retention capacity and metabolic control by the canopy. Establishing native vegetation to fulfil broader biodiversity objectives whilst retaining willows for their potentially positive roles is a management challenge. In principle, establishing native species on the upland-sides of fringing willows and under willow canopies will direct succession toward a preferred outcome without destructive disturbance. Understanding of when benefits of willows outweigh their costs is a notable knowledge gap.

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Statement of Authorship

Except where explicit reference is made in the text of the thesis, this thesis contains no material published elsewhere or extracted in whole or in part from a thesis by which I have qualified for or been awarded another degree or diploma. No other person's work has been relied upon or used without due acknowledgment in the main text and bibliography of the thesis.

Michael Wilson

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Table of Abbreviations

AFDW	ash free dry weight
BOM	benthic organic matter
CBOM	coarse benthic organic matter
CR_{24}	24 hour community respiration
DO	dissolved oxygen
DOC	dissolved organic carbon
DOM	dissolved organic matter
FPOM	fine particulate organic matter
GPP	gross primary production
LOI	loss on ignition
LWD	large woody debris
NDM	net daily metabolism (daily GPP - CR_{24})
NPP	net primary production
OBA	Ordovician bedrock aquifer
PAR	photosynthetically active radiation
P/R	ratio of GPP to CR_{24}
QBA	Quaternary basalt aquifer
R_A	autotrophic respiration
R_H	heterotrophic respiration
SMI	stream metabolism index
T_P	BOM turnover based on processing
T_{PT}	BOM turnover based on processing and transport
SaCrM	Sailors Creek mixed site
SaCrN1	Sailors Creek native site 1
SaCrN2	Sailors Creek native site 2
SpCrW	Spring Creek willow site
WoCrN	Wombat Creek native site
WoCrW	Wombat Creek willow site

INTRODUCTION

Problem statement

All indigenous riparian trees of mainland south eastern Australia are evergreen. Since the arrival of Europeans deciduous exotic species have colonised some riparian zones. In Victoria, willows are one such group of species. They have been planted during river management works due to their effectiveness in stabilising eroding banks and channels. Willows have spread both vegetatively and by seed from these and landscape plantings (Cremer 1995) and the extent of willow spread along coastal and inland streams and rivers has precipitated widespread concern over possible ecological effects to both riparian and stream zones (Frankenberg 1995, Schulze and Walker 1997, Ladson *et al.* 1997). Willows, with the exception of *Salix babylonica*, *S.* x *calodendron* and *S.* x *reichardtii*, have been named weeds of national significance (National Weeds Strategy Executive Committee, 2000). Victorian catchment management authorities and landholders are currently engaged in extensive willow removal (Ladson *et al.* 1997). Removal programs are commonly supported by public grants and are strongly encouraged by government and private catchment management organisations.

In forested small order streams, riparian litter fall is frequently regarded as the major contributor of organic matter, which, in turn, underpins the energetics of the stream ecosystem (Fisher and Likens 1973, Mulholland 1981, Benfield 1997, Wallace *et al.* 1997, Webster *et al.* 1997). It has been argued that a change in the timing or quality of litter fall resulting from a change in riparian vegetation phenology would affect stream energetics and be reflected in the composition of the stream biota (Campbell 1993, Schulze and Walker, 1997). This suggestion has some empirical support in that differences have been observed in the magnitude and seasonality of shading by a willow canopy compared to a red gum canopy (Besley 1992) and in the in-stream decomposition rates of willow and eucalypt leaves (Pidgeon and Cairns 1981, Yeates 1994, Schulze and Walker 1997). However, Australian studies that have directly compared the biota within willow and native tree lined stream reaches have been inconclusive (Pidgeon 1978, Besley 1992, Hardwick *et al.* 1995, Read and Barmuta 1999, Schulze and Walker 1997), with only Pidgeon (1978) and Read and Barmuta

(1999) showing notable differences. The streams and rivers in which these studies have been conducted span a range of orders, climates and riparian vegetation types, so generalisations that can be made are few. Many of these studies are confounded by the extent of catchment dysfunction from other degrading processes but deciduous riparian vegetation, despite the fact that it is a novel element in SE Australia, has not resulted in a novel stream biota under all conditions.

Pidgeon (1978) observed a shift from autotrophy to heterotrophy and lower invertebrate biomass as a stream flowed from open woodland and pasture lined reaches into a willow lined reach. His work is widely cited in management literature as a comparison of native lined streams and willow lined steams (Frankenberg 1995, Ladson *et al.* 1997). However, the autotrophic woodland reach appeared degraded with no understorey, a very sparse canopy and direct access to the stream by grazing animals. Streams flowing through native forests with dense riparian canopies appear to be heterotrophic (Chessman 1985, Treadwell 1995, Davies and Bunn 1999). Pidgeon's study may be best considered an illustration of the energetics of degraded stream reaches compared to a willow lined reach. Factors confounding the interpretation of willow-induced changes to the biota include lack of replication and stock access to non-willow sites.

Read and Barmuta's (1999) study, whilst finding biological differences between sites, has similar constraints with all native lined reaches studied in low disturbance forested catchments upstream of high disturbance willow lined reaches adjacent to pasture and orchards. The component of variation in biota due to willow presence cannot be separated from the component due to catchment clearing and landuse difference. Besley (1992), Hardwick *et al.* (1995), and Schulze and Walker (1997) all concluded that there were few differences in the bank biota of willow and native tree lined reaches on mid- to large order rivers (>6^{th} order) in the Murray-Darling Basin. These latter studies all concerned *Salix babylonica*, whilst Pidgeon (1978) and Read and Barmuta (1999) considered *S. fragilis*. These species do differ in habit and may differ in characteristics such as the timing, quality or quantity of litterfall. They are all deciduous and are likely to have more in common than between them and native evergreen vegetation.

The suggestion that canopy and phenological differences between deciduous willows and evergreen native riparian vegetation would lead to differences in the biota in adjacent stream reaches has not been confirmed. One possibility is that the role of riparian litter fall in stream organic matter dynamics is less than would be expected from interpretation of overseas studies. There are few data, with only one published organic matter budget for an Australian catchment (Treadwell *et al.* 1997). This study suggested that groundwater was the major contributor of organic matter (75% of total inputs) to streams in a central Victorian catchment with riparian litter contributing only 15% of the total. If such a pattern is generally applicable in similar Victorian streams then phenological influences on litterfall need to be discussed in the context of total organic matter dynamics.

In attempting to draw a generalised model of the impact of willows on Australian streams and rivers, the breadth of scale (both temporal and spatial) in the above studies is as much a confounding factor as the range of conclusions. Whilst willows are exotic to all of Australia, lotic systems differ markedly in physical, hydrological and biological characteristics across the continent (Lake 1995). Given this variability, site specific tools for evaluation of willow impacts are potentially more useful than general prescriptions, particularly at a national level. The exploration of partial organic matter budgets as one such tool was an underlying aim of the present study.

Aims

Fisher and Likens (1973) pioneered the use of organic matter budgets to integrate inputs, storages and outputs of organic matter. Budgets identify the known and potential sources and fates of energy and materials in an ecosystem and quantify their relative magnitude (Minshall 1996). Cummins *et al.* (1983) concluded that organic matter budgets could be effectively used in small-scale research to compare selected processes between diverse stream ecosystems. The present study stems directly from this perspective and can be regarded as an alternative approach to the inconclusive studies of biota under willow and eucalypt lined streams.

The application of partial organic matter budgets to aquatic ecosystem management and research has been limited to impacts of logging (Meyer and Tate 1983, Webster *et al.* 1990) and beaver presence-absence (Naiman *et al.* 1986). No studies have attempted to apply this tool in Australia. The present study uses a potentially generic tool (the partial organic matter budget) to explore a specific aquatic ecosystem issue. The colonisation of southern hemisphere riparian zones by exotic deciduous species (e.g. willow colonisation in New Zealand: Collier and Winterbourn 1986, Glover and Sagar 1994, Lester *et al.* 1994a, 1994b, 1995 and South Africa: Brown and Gubb 1986, Henderson and Wells 1986, Immelman 1987, Henderson 1989, 1991), and the converse colonisation of northern hemisphere riparian zones with evergreen exotic species (e.g. Pozo *et al.* 1997) gives this study global relevance. If conclusions from partial organic matter budgets can identify critical areas of research and management for a particular site and issue, then resources may be more effectively focussed.

A postulate underlying the present study was that phenological influences on litterfall would be a significant factor in stream organic matter dynamics within low order forested stream reaches lined by deciduous and evergreen riparian vegetation. Based on the axiom that willows are deciduous and the native riparian vegetation of the study site is evergreen, we can postulate that seasonality of litterfall would be a key distinguishing feature (e.g. Campbell 1993). However, the consequence of this postulate for overall organic matter dynamics in a stream reach requires understanding of the influence of riparian vegetation type on all other organic matter inputs and storages.

Additionally, the present study explored previously unquantified observations that willows were capable of exerting a strong influence on stream retention characteristics. The efficacy of willows in controlling stream erosion is well acknowledged (Crouch *et al* 1987, Ladson *et al*. 1999) and implies a strong geomorphological role including effects on hydraulic resistance and sediment retention capacity. The way materials, including organic matter, are retained, stored and transformed is at the heart of stream ecology (Winterbourn and Townsend 1991, Fisher *et al.* 1999). Exploration of the influence of willows and other riparian

vegetation on organic matter standing crop and aggradation can expand the understanding of willow impacts on stream ecosystems.

To explore these postulates, partial organic matter budgets were prepared for stream reaches lined by willows and by native vegetation. Chapter 2 reviews the history and ecological consequences of willow spread in Australia as well as past research and current theory surrounding stream organic matter in the context of south east Australian streams. Chapter 3 describes the location, physical characteristics and land use history of the study site. Chapters 4 to 7 are experimental chapters and present methods, results and discussion for experimental investigations of organic matter inputs, respiratory output and storage. Chapter 4 concerns litter fall from riparian vegetation, Chapter 5 the inputs of dissolved organic matter via groundwater, Chapter 6 the photosynthetic inputs and respiratory outputs from community metabolism and Chapter 7 the storage of organic matter and sediment in the stream benthos, large woody debris and debris dams. The results from the four experimental chapters are compiled in Chapter 8 to form partial stream organic matter budgets for willow and native forest lined reaches as well as the study catchment as a whole These budgets are critically appraised and conclusions for both ecological research and willow management are presented in Chapter 9.

REVIEW OF THEORY

Introduction

Understanding of linkages between riparian zones and surface streams has been central to advances in running water ecological theory (e.g. river continuum concept, Vannote *et al.* 1980; ecotones, Naiman *et al.* 1988; the telescoping ecosystem model, Fisher *et al.* 1998). Similarly, current stream conservation and management has emphasised riparian-stream linkages (Cummins 1992, Bunn *et al.* 1993, Rutherfurd and Bartley 1999). Theoretical and applied aspects of riparian-stream linkages blend with conservation biology and invasion ecology when considering willow (*Salix* spp.) spread in south eastern Australia. Willows are exotic to Australia but have had widespread use in bank stabilisation on stream and river systems. They have spread along watercourses and are the focus of widespread management activity (Ladson *et al.* 1997). In response, willows, with the exception of *Salix babylonica, S.* x *calodendron* and *S.* x *reichardtii,* have been named weeds of national significance (National Weeds Strategy Executive Committee, 2000). Little research has been conducted yet it has been widely suggested that fundamental stream ecological processes may be affected by willow spread (Campbell 1993, Frankenberg 1995, Ladson *et al.* 1997). This review will discuss lotic ecological research that has focused on exotic willow spread. In addition, literature relating to the study of organic matter dynamics in streams lined by willow and native forests will be reviewed to establish the theoretical basis for the subsequent chapters in this thesis.

Willow taxa and history of introduction

More than 100 species or varieties of willow have been introduced to Australia (Cremer 1995) with subsequent hybrids arising in Australia (Ladson *et al.* 1997). Most were deliberately introduced for utilitarian uses including stream stabilisation, timber, channel marking, basketry, horticulture and ornament (Carr 1995, Frankenberg 1995, Schulze and Walker 1997). Twenty-three taxa are reported as naturalised in Victoria with four regarded as seriously invasive, *Salix fragilis* var. *fragilis, S.* x *rubens, S. cinerea* and *S. nigra* (Ladson *et al.* 1997).

Most reports date willow introduction no more accurately than 'early' in European settlement of Australia (e.g. Frankenberg 1995, Smith and Starr 1999). It has been suggested that whilst ornament and familiarity motivated some planting (Frankenberg 1995), catastrophic anthropogenic channel change initiated widespread willow establishment in NSW between 1850 and 1870 (Smith and Starr 1999). The extent of stream metamorphosis has been extensively documented (Brizga and Finlayson 1990, Burston and Good 1996, Fisher 1996, Brooks 1999a, Nanson and Doyle 1999, Starr 1999, Vincin 1999) and, in many cases, appears to have been initiated within the first years or decades following catchment clearing (Nanson and Doyle 1999, Starr 1999). For Europeans, willows would have been familiar features of riparian landscapes and, presumably, planting them represented a culturally appropriate response to channel deepening, widening and erosion. The intersection of channel morphology change and willow colonisation is further explored in the next section (Section 2.3).

Willow establishment subsequently became an accepted and widespread tool in river management works planned or conducted by government agencies (Warner and Bird 1988). This practice has continued until present (Crouch *et al.* 1987, Ladson *et al.* 1997) albeit on a much smaller scale than in the past. Victorian river management authorities were surveyed by Ladson *et al.* (1997) to gauge current attitudes towards willows along waterways. Less than 200 willows per year were being planted and nearly all authorities were engaged in willow management works (lopping, poisoning or removal). Willows are currently seen as a tool of last resort with an emphasis on use of sterile non-brittle hybrids with low regeneration potential (Ladson *et al.* 1997).

Establishment of new populations has not always been a human mediated process. Vegetative propagation has spread willow downstream from plantings and seed has blown laterally and floated down watercourses and within wetlands (Cremer *et al.* 1995). Naturalised populations from seed are thought to be a very recent phenomenon in Australia with the oldest confirmed seedling populations dated to approximately 1970 (Cremer *et al.* 1995). It is considered an important emerging management issue (Ladson *et al.* 1997, Riddell 1997). Willows are estimated to occur on 30 000 km of Victoria's river frontage (Ladson *et al.* 1997). With a total river frontage of 68 000

km, willows represent a dominant and potentially important component of many stream corridor ecosystems.

Exotic willows are spreading in riparian zones in New Zealand and South Africa, and both countries have a growing scientific and management literature on the issue (New Zealand: Collier and Winterbourn 1986, Glover and Sagar 1994, Lester *et al.* 1994a, 1994b, 1995. South Africa: Brown and Gubb 1986, Henderson and Wells 1986, Immelman 1987, Henderson 1989, 1991). The concerns of scientists and managers in these two countries are similar to those in Australia. Scientific understanding and techniques of control are being shared, especially between Australia and New Zealand (pers. com. H. Tane, Centre for Catchment Ecology, Twizle, 1999).

The scale of willow distribution and a historical shift within river management authorities from a resource-engineering management paradigm toward a watershed ecosystem (integrated catchment) paradigm (Healey 1998) are factors that have led to debate over appropriate responses to willows within academic, management and broader communities. From a southern NSW perspective Smith and Starr (1999) called the willow debate "possibly the most contentious issue in river management" and Bobbi (1999) stated that community groups in Tasmania have "made it clear" that control and management of willow infestation is the most important river management issue. Identifying and communicating the contribution of such synthetic vegetation (admixtures of native and exotic species) to biodiversity and landscape stability was a challenge identified by Bridgewater (1990).

Ecological and geomorphological impacts of willows

A view has emerged that willows cause a range of detrimental impacts (Frankenberg 1995, Ladson *et al.* 1997, Bobbi 1999, Smith and Starr 1999). These, it is claimed, can be readily observed from field observation (Frankenberg 1995) but studies are few. There are three studies published in peer reviewed journals that have directly investigated willow-stream interactions in Australia (Pidgeon and Cairns 1981, Cremer *et al.* 1995, Schulze and Walker 1997). In addition, there are a number of unpublished postgraduate (Pidgeon 1978, Hardwick *et al.* 1995, M. Read, University of Tasmania pers. comm. 1999) and undergraduate theses (Besley 1992, Lindberg

1992, Romer 1994, Yeates 1994). Numerous management reports (reviewed in Ladson *et al.* 1997) and articles from conference proceedings (Campbell 1993, Frankenberg 1995, Carr 1995, Bobbi 1999) speculate on willow impacts on stream ecosystems and suggest methods of control or removal.

Research has been dominated by comparisons of willow and native leaf pack decomposition and colonisation or field surveys of aquatic biota at willow lined and native forest lined reaches (Pidgeon and Cairns 1981, Hardwick *et al.* 1995, Besley 1992, Schulze and Walker 1997, M. Read, University of Tasmania pers. comm. 1999). Pidgeon's (1978) energetic study of *Eucalyptus* woodland, pasture and willow lined reaches on a NSW stream is the only one to directly investigate ecosystem processes. Pidgeon concluded that shading and allochthonous inputs were significantly greater at a willow lined reach than at pasture or woodland lined reaches. Seasonal variation in canopy gap fraction was greatest at willow sites. The season of greatest canopy cover by willows coincided with the season of greatest incident photosynthetically active radiation (PAR) and the season of least cover coincided with least incident PAR, resulting in lower seasonal variation in total PAR recorded at the stream bed. The relatively dense shade at the willow site was considered the dominant factor explaining observed lower rates of gross primary production (GPP 30-50% of other sites). Metabolically the willow reach was heterotrophic with rates of GPP less than community respiration (P/R ratio<1) and the open and woodland reaches autotrophic (P/R>1).

Incident PAR was also reduced by up to 80% by a willow canopy shading New Zealand streams (Lester *et al.* 1994b). In a pattern similar to that observed by Pidgeon (1978) seasonality of PAR measured at the stream surface was lower at willow sites than open sites. Both these studies showed that even in winter, willow canopies still reduce incident PAR by at least 50%. Despite the differences in light regime observed at willow and open sites on the New Zealand streams there were no differences in periphyton chlorophyll a. This was attributed to increased periphyton turnover associated with grazing pressure from larger numbers and biomass of macroinvertebrates, rather than lower photosynthetic efficiency of open site periphyton.

Dense shade has been considered an essential feature of upland stream restoration precisely because it controlled the community structure of aquatic autotrophs (Davies and Bunn 1999). Studying the Mary River in Queensland, these authors noted that in-stream GPP switched from palatable microalgae to filamentous algae and/or macrophytes as light levels increase. A decline in stream health was stated to occur when GPP exceeded community respiration and the stream became autotrophic. Their modelling of the Mary River catchment predicted this would occur at <40-50% canopy cover. Regulation of undesirable macrophytes by willow canopies has had a positive impact for stream management in Tasmania (Bobbi 1999).

Decreased autotrophic carbon inputs that resulted from shading were potentially compensated for by increased inputs of allochthonous organic matter at Pidgeon's (1978) study site. Annual total litter accession to the stream at the willow site was eight times that at the woodland site. The seasons of greatest and least litter accession were autumn (62% of annual total) and winter (5%) respectively at the willow site and spring (48%) and autumn (12%) at the woodland site. However, the willow site had lower macroinvertebrate biomass and production, leading Pidgeon to the conclusion that willow litter was unpalatable to the macroinvertebrates at the site and thus contributed little to stream energy flow. When subsequently testing this hypothesis Pidgeon and Cairns (1981) found that willow leaves were rapidly consumed and concluded that differences in macroinvertebrate productivity between reaches lined with willow and native vegetation could not be explained by refractory exotic litter. This is supported by a stable carbon isotope study in New Zealand that showed allochthonous derived carbon could contribute 8-74% of late instar insect larvae body carbon at willow sites compared to 0-23% at open sites (Lester *et al.* 1995). Further Australian and New Zealand studies have concluded that willow litter is palatable, can be a preferred food source for aquatic macroinvertebrates and can play a major role in aquatic ecosystem energy flow (Collier and Winterbourn 1986, Lester *et al.* 1994a, Yeates 1994, Schulze and Walker 1997).

The dominant focus of academic speculation on willow-stream interactions has remained on the phenology of the canopy (Campbell 1993, Frankenberg 1995).

Timing and quality of litterfall, with consequential impacts on aquatic food webs, has been suggested as a key feature that would distinguish deciduous willow lined reaches and evergreen native forest lined reaches. Campbell (1993) compared studies of litter accession to streams lined with deciduous forests in the northern hemisphere with those lined by evergreen forests in south eastern Australia. Data sets were small (n=3 to n=14) and the intention of the article was to propose research questions. Key findings and speculations included that amounts of litter accession were similar but that composition and seasonality were different. A higher proportion of bark and a lower proportion of leaves may fall into Australian forested streams and the season of peak litter accession was summer in Australia and autumn in the northern hemisphere. In addition the proportion of litterfall entering streams in the season of peak litter fall and season of minimum fall were ~40% and ~15% respectively in Australia and ~70% and ~5% in the northern hemisphere. This pattern is supported by Pidgeon's (1978) study. Despite the intervening 15 years, Campbell concluded that, from a management perspective, there are conspicuous gaps in our knowledge of the consequences for organic matter dynamics of replacement of native riparian vegetation with deciduous exotic species such as willows.

Frankenberg (1995) and Ladson *et al.* (1997), in response to a paucity of research, took similar speculative approaches to willow impacts on riparian and aquatic environments in south eastern Australia. They suggest that willows reduce macroinvertebrate numbers in streams, citing Pidgeon (1978) and Besley (1992). However, Besley showed no significant differences between root habitats of *Casuarina cunninghamiana, Eucalyptus camaldulensis* and *Salix* spp. in winter and no difference between *C. cunninghamiana,* and *Salix* in autumn. Root habitats of all tree species supported greater macroinvertebrate diversity but similar numbers when compared with bare bank habitats. Besley concluded that "the null hypothesis that there is no difference between native and exotic tree species is accepted based on autumn and winter results". Read and Barmuta (1999) obtained similar results during winter and autumn in Tasmania but in summer showed high disturbance willow sites to have lower density and biomass of macroinvertebrates than low disturbance native and rainforest sites. Studies of macroinvertebrate colonisation of willow and native

leaf packs are similarly inconclusive (Pidgeon and Cairns 1981, Schulze and Walker 1997).

A reason for the difficulty in elucidating willow impacts on stream biota is lack of sites that can be considered 'controls'. As earlier stated (Section 2.2), river metamorphosis has occurred in a large number of south east Australian catchments since the time of European settlement. Changes to the fluvial landscape have coincided with the spread of willows and finding sites unaffected by impacts other than willow invasion is difficult. In addition, Schulze and Walker (1997) question whether the scale of willow colonisation of some catchments may be so great as to effectively mask influences by relatively isolated native riparian vegetation communities on macroinvertebrates. Finding control sites that are characteristic of pre-disturbance native riparian vegetation and associated stream biota is a major challenge This is particularly true for lowland river reaches that are degraded by the cumulative impacts of damaging land use practices throughout their catchments. Direct measurement of material or energetic linkages between native riparian vegetation and lotic systems may be compared to those of nearby willow lined reaches where these measures are not strongly influenced by fluvial metamorphosis or other widespread impacts. For example, organic matter accession, light penetration of the canopy, large woody debris (LWD) delivery, vegetation contribution to channel roughness and root contribution to bank strength could be measured in isolated patches of native vegetation in highly modified landscapes. Circumstantially support for Schulze and Walker's (1997) postulate of widespread influences of willows on biota might be provided if energetic and material dynamics were dramatically different between native and willow reaches but the biota was similar.

In-stream LWD derived from willows has been thought to be shorter lived, of poorer habitat value and less frequently recruited compared to that of *Eucalyptus camaldulensis* (Frankenberg 1995, Ladson *et al*. 1999). Willow wood is less dense and less durable than wood from the most common *Eucalyptus* and *Acacia* species in riparian communities of south eastern Australia (Bootle 1983). Unfortunately, there are no published studies of recruitment of LWD from willows. Native fish numbers in the Ovens River, Victoria have been shown to increase by a factor of six because of

willow debris accumulation at a previously open site (Koehn 1987). Glova and Sagar (1994) found trout abundance and biomass to be greater in willow lined reaches than open reaches of three New Zealand streams. Eel abundance in one stream and biomass in another were greater in willow reaches but distribution of other fish was not related to willow presence or absence. M. Read (University of Tasmania, pers. comm. 1999) found reduction in LWD to correspond with number and size of some fish species. Active and historic removal of in-stream LWD accounted for the distribution of LWD in the rivers studied rather than recruitment or retention characteristics of riparian vegetation. Willow LWD supported a similar community of macroinvertebrates but a lower density when compared to native LWD and Read concluded that willow LWD was a poor ecological substitute for complex native LWD.

All M. Read's (University of Tasmania, pers. comm. 1999) willow sites were within cleared agricultural landscapes and were downstream of native sites within low disturbance forests. Catchment clearing and resulting impacts on LWD have been shown to impact on organic matter export and turnover (Webster *et al.* 1990, Bird and Kaushik 1992), sediment and nutrient delivery and retention (Lake and Marchant 1990) and macroinvertebrates (Winterbourn and Townsend 1991, Bird and Kaushik 1992). Read acknowledged that attributing changes in aquatic fauna to willow presence or to intrinsic characteristics of willow LWD when catchment land use, riparian disturbance history, and river management practices also differ between paired sites is problematic.

Dissolved oxygen concentrations in willow lined reaches of a Tasmanian stream were lower than in adjacent native vegetation lined reaches during summer (low flow) but differences were small in winter and spring (Bobbi 1999). Organic matter retention within willow root mats and reduced autotrophic production as a result of dense canopy cover were suggested causes of the observed pattern. Temporary streams in *Eucalyptus* forests showed similar patterns of summer organic matter accumulation and dissolved oxygen decrease as flow dwindled (Boulton and Suter 1986). In autumn the return of flow caused dissolved oxygen levels to rise sharply even though they accompanied a pulse of litter. Both surface stream flow and organic matter

influenced dissolved oxygen levels. From these two studies we can postulate that when low flow and large organic loadings coincide, dissolved oxygen will be depressed under either willow or eucalypt canopies. Combined hydrological and organic matter dynamics data would allow us to predict when large organic matter loadings are likely to coincide with low flows. Litterfall studies under willow canopies would improve such predictions.

Documentation of geomorphological consequences of willows on Victorian streams is dominated by reports commissioned by management authorities (numerous citations in Ladson *et al.* 1997). The efficiency of willow root systems in armoring banks and trapping sediment has been seen as contributing to reduced bank and bed erosion. The ability of willows to establish in mid-channel, grow rapidly, resist high flows and accumulate sediment at their bases has been implicated in reducing channel capacity (Brizga and Finlayson 1990), increasing flooding, causing channels to become wider and shallower and increasing braiding (Ladson *et al.* 1997, Ross 1994). Stream 'choking' has been a common motivation for willow removal programs (Cremer *et al*. 1995, Frankenberg 1995, Thomas 1995, Thexton 1995, Ladson *et al.* 1997, Bobbi 1999, Outhet *et al.* 1999). Retention of fine sediment leading to decrease in mean substrate particle size and willow root mat occupation of interstitial spaces has been considered detrimental to macroinvertebrate diversity (Lester *et al.* 1994b, Frankenberg 1995). Retention of organic matter in willow lined reaches has been considered a form of 'organic pollution' leading to dominance by deposit feeding collectors (P. Suter, La Trobe University, Wodonga, pers. comm. 1996).

The intersection of willow colonisation and change to fluvial landscapes in south eastern Australia can be illustrated by examples. Brizga and Finlayson (1990) studied channel avulsion, and subsequent river metamorphosis, that occurred on the Thompson River in 1952. Of particular interest is the observation that in the 1940's the Thompson River at the location of the avulsion was 'overgrown with willows and choked with snags'. Such an association can tempt observers to search for a causal relationship between willows and channel change (e.g. Rutherfurd 2000), but in this case, Brizga and Finlayson (1990) made no direct comment on the cause of the avulsion. The old channel was subsequently cleared of willows and snags, to achieve

an increase in channel capacity to aid its function as a supply channel for irrigation and stock watering. The new channel was formed in farmland approximately 2.5 km away from the old channel and its associated riparian vegetation. Without intervention, colonisation of the riparian zone of this new channel is constrained by the propagules delivered from upstream riparian zones that were still dominated by willows in 1990. Current management frameworks (e.g. Ladson *et al*. 1997, Askey-Doran 1999) would advocate removal of any successful willow colonisation and planting of indigenous riparian species, an action that contrasts the use of willows last century in response to major channel change in NSW (Smith and Starr 1999).

Andrew Brooks and Gary Brierley have described channel metamorphosis in the Bega Valley (Brooks 1999a, Brierley 1998, Brooks and Brierley 2000). Three phases of post-European channel evolution were proposed. Phase 1, 1850-1926, was characterised by channel expansion by up to 340% and an increase in sediment supply. This was largely attributed to anthropogenic causes including riparian vegetation removal, drainage and stock damage, not to natural phenomena such as flood frequency or flood magnitude. Phase 2, 1920-1960, was a quiescent phase with little gross morphological change to the lower Bega River channel. Phase 3, 1960-present, was a period in which channel capacity was reduced in association with willow colonisation within the channel. Brooks and Brierley (2000) suggest channel capacity has been reduced by 50% at some locations and by an average of 25% across their entire study reach. Aggradation by willows was accelerated when ground cover, including exotic grasses and *Artemisia verlotorum*, colonised aggrading sediments. They make the point that the relationship between vegetation colonisation within a channel and morphological change is not linear. Rather, vegetation modification throughout the catchment and the riparian zone ‘pre-condition’ a channel to adjustments during subsequent floods.

A final example comes from the Goulburn (Mid-Goulburn Catchment Coordinating Group, 1993) that followed preparation of a report on the fluvial geomorphology of the Goulburn River basin (Erskine *et al.* 1993). As for the lower Bega River in the previous example, three phases of channel morphology change since European settlement were identified for the tributaries of the Goulburn River. During the

depositional phase (1880 to 1889), large amounts of sediment were released to the tributaries and deposited on the pre-existing floodplain. During the incising phase (1900-1950), tributary incision lead to channel bank erosion and sediment transfer to the lower reaches of the Goulburn River. Finally, in the present phase, the worst instabilities had passed, with most streams having returned to a state of relative stability. The evidence included the fact that most gully heads had been stabilised by willow roots, bedrock bars or by a flattening of channel grade. Channel incision would begin again if these controls were outflanked. For the tributaries of the Goulburn River the key emerging management issue identified was dryland salinity, which had the potential to destroy riparian vegetation and thus initiate new bank erosion.

The stabilising role of riparian vegetation, particularly willows, in the incised tributaries of the Goulburn River was contrasted with the role of willows in reaches downstream of Lake Eildon (Erskine *et al.* 1993). In the latter reaches, the geomorphology of the channel was determined by Lake Eildon, which had reduced sediment loads and flood frequency and magnitude. The consequences of this flow regulation included willow colonisation of fine-grained depositional sites within the channel. The concerns raised were that, whilst riparian vegetation was essential for channel stability, willow invasion could block the channel and thus cause bank erosion, debris dams and avulsions. It was recommended that willows be selectively replaced with native vegetation to avoid problems of channel instability, reduced biodiversity and deteriorating habitat (Erskine 1993). Implicit in this recommendation is that ‘native vegetation’ would not totally block the channel, an acknowledgment, in a general sense, that willow and native vegetation have different hydraulic retention capacity. Erskine (1996) considered the regulation of the river (Lake Eildon) a first order impact and willow invasion a subsequent or fourth order impact, predicated on the river regulation.

At some locations geomorphologists wish to see increased sediment and organic matter retention, increased floodplain-stream connectivity and restoration of discontinuous pools and ponds characteristic of pre-European upland streams (Brierley 1998, Brooks 1999a, 1999b, Cohen 1999, Doyle *et al.* 1999). In the

example of the Goulburn River above, willow invasion was seen as potentially destabilising in the mid-reaches of the river primarily because of its capacity to block the channel. Regulation of the river had modified flood and flow regimes (Erskine *et al.* 1993) meaning an alteration in the channel form and a decrease in the floodplain-river connectivity through lower flood frequency and magnitude. The build up of debris dams and blocking of the channel by willows would suggest potential for decreased material export (e.g. Bilby 1981) and increased river-floodplain connections during flows that would otherwise have been carried within the channel. Much management debate for river systems focuses on conflicting views, where advantages and disadvantages can be identified for the same characteristic, be it retention capacity, shade, debris dams or floodplain-river connectivity.

Riparian systems are expected to perform an increasing number of ecological and social functions (Naiman and Décamps 1997) and it is perhaps inevitable that conflicts will arise between competing functions and no single riparian system will be able to perform all these functions (Naiman and Décamps 1997). The conflicts illustrate the difficulty of approaching a natural resource issue through the study of only isolated components. It has been suggested that the flows of energy (e.g. food chains and trophic processes) and materials (e.g. nutrient cycles) are essential elements in a more holistic approach to analysis of ecosystem function (Jørgensen 1997, Dickinson and Murphy 1998). Pidgeon's (1978) study of stream reaches illustrates how an energetic approach can be applied to the issue of willow colonisation of Australian riparian zones. The present study analyses material flows in willow lined streams with the aim of furthering the holistic understanding of willow colonisation in Australia.

Organic matter budgets

The general design of organic matter budgets is based on a mass balance approach in which inputs (= imports), standing crops (= storages) and outputs (= exports) are quantified for a stream reach or catchment (Minshall 1996, Webster and Meyer 1997b).

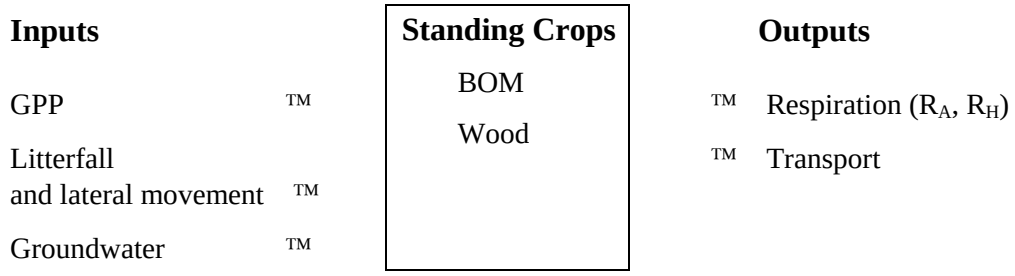


Figure 2.1 The components of a stream organic matter budget, from Webster and Meyer 1997b. Abbreviations are given in Table of Abbreviations.

A budget produced in this way illuminates and quantifies the magnitude of sources and fates of organic matter. They are often inseparable from analysis of energy flow (Minshall 1996). This is largely because organic matter is a key component of both the material cycling and energetic subsystems of stream ecosystems (Dickinson and Murphy 1998). Organic matter budgets and watershed budgets for individual ions have informed influential insights into the structure and function of flowing water (Cummins *et al*. 1983). The pioneering budgets at Hubbard Brook (Fisher and Likens 1973, Bormann and Likens 1979) contributed to the realisation that small streams in forested catchments are heavily dependent on energetic and nutritional resources of terrestrial origin. The understanding that terrestrial dependence decreased with increasing channel size underpins the River Continuum Concept (Vannote *et al.* 1980). The finding that aquatic system dependence on terrestrial resources may vary with riparian vegetation type (Minshall 1978), was developed through organic matter budgets.

Stream organic matter budgets have been used to evaluate impacts on aquatic ecosystems from catchment and riparian modification, for example logging (Meyer and Tate 1983, Webster *et al.* 1990) and beaver presence-absence (Naiman *et al.* 1986). The only published Australian stream organic matter budget is for Keppel Creek, a fourth order forested stream in central Victoria (Treadwell *et al.* 1997). Inputs were dominated by groundwater DOM (an estimated 75% of total inputs). This is the largest proportion attributed to groundwater for any published organic matter budget (Webster and Meyer 1997a). If true for central Victorian streams then GPP

and litterfall could play a much smaller role in organic matter dynamics in these streams than would be inferred from overseas studies. Consequently, a change in riparian forest from evergreen to deciduous may have little overall impact and thus not be reflected in changes in biological communities. Treadwell *et al.*'s (1997) study provides impetus for detailed study of organic matter dynamics in Australian streams, particularly those quantifying groundwater DOM contribution.

Problems in the evaluation of stream organic matter budgets have been recognised (Cummins *et al.* 1983, Webster and Meyer 1997). Many problems stem from the desire to compare ecosystem functional properties within and between biomes (Cummins *et al.* 1983). Such comparisons can be rendered misleading, or at least ambiguous, through the temporal and spatial variation inherent in those data necessary for mass balance budgets. Webster and Meyer (1997) suggested that differences between published organic matter budgets reflect what was measured, how it was measured and what was left out, rather than differences in biological function between the lotic systems.

A fundamental problem in existing budgets is that they assume steady state conditions in parameters or have quantified a range of values for parameters over time frames shorter than those known to have an influence (Cummins *et al.* 1983). For example, flood events of increasing magnitude and decreasing frequency will scour deeper sediments, extend further onto the floodplain and reorganise larger debris dams (Cummins *et al.* 1983). Annual organic matter budgets will record conditions reflecting the hydrological history of the site. Each site will be variably situated along a recovery trajectory, as the site adjusts to its disturbance history (Cummins *et al.* 1983, Fisher *et al*. 1998). Assumptions of steady state during any given year are probably not valid (Cummins *et al.* 1983).

Further problems in the interpretation of stream organic matter budgets lie in the fact that they reflect physical transport and storage characteristics more than metabolic processes (Cummins *et al.* 1983). The problem here lies in interpretation, rather than there being no value in discussing material transport and storage. Historically, when authors have searched for metabolic answers for budget differences between streams,

they have avoided confronting the complexity of organic matter transport, retention and deep storage pools in hyporheic and floodplain locations. Cummins *et al.* (1983) suggested that further understanding of ecosystem properties would be obtained through improved understanding of organic matter storage dynamics. Fifteen years later this intellectual challenge has been embodied in the Telescoping Ecosystem Model (Fisher *et al.* 1998).

. If the primary objective of full organic matter budgets is to determine ecosystem functional properties within and between biomes then Cummins *et al.* (1983) saw this being achieved with long term data sets and improved monitoring of storage dynamics at selected sites. Small scale studies "should focus on comparisons of selected processes between diverse stream ecosystems rather than attempt to determine whole system budgets" (Cummins *et al.* 1983). This approach was undertaken by both the present study and by Webster *et al.* (1990). The latter had a primary objective to measure inputs, outputs and storage of organic matter in logged and reference streams but they also used these data to produce organic matter budgets. They quantified the effects of logging from a budget perspective and suggested that many of the major problems identified by Cummins *et al.* (1983) did not apply to their comparative study.

Firstly, Webster *et al.* (1990) assumed that the neighbouring streams in their study had similar climatic histories, thus addressing Cummins *et al.*'s (1983) major concern with comparisons between divergent stream systems and biomes. Short-term budgets evaluating anthropogenic disturbance were thus undertaken on streams that were at a similar successional state, in relation to flood recovery and other major disturbances. They found some evidence that even close streams behaved differently to the same storm event in relation to particulate export, however this was not an aspect investigated in the present study. In addition, the history of past events was carefully considered, as their study was part of a long-term investigation of forest hydrology. Secondly, they paid particular attention to detrital storage in their sampling regime, as this parameter had been noted to be poorly quantified in most previous budget studies. They noted statistically significant differences between streams, confirming that their experimental design had the necessary statistical power (Cohen 1988). They

acknowledged that deep storage in some areas was probably not adequately sampled. Thirdly, steady state was not assumed and all budget parameters were either measured directly or estimated from measured data. Finally, in attempting to overcome Cummins *et al.*'s (1983) concern that methods for measuring particulate export are often inadequate, Webster *et al.* (1990) sampled on an event basis at frequencies as high as every 5 minutes. They developed empirical models utilising the long hydrological record and data from all storm events sampled. Despite this, they concluded that there remained substantial problems in estimating annual particulate export in streams with highly variable hydrology.

General approach for the present study

The general approach for this thesis arises from Cummins *et al.*'s (1983) and Webster *et al.*'s (1990) perspectives. Partial organic matter budgets were constructed for stream reaches lined with willows or native forest. The term 'partial organic matter budget' is deliberately used in acknowledgment that particulate and dissolved organic matter export were not quantified in the present study. The organic matter budget components considered were inputs from litterfall, groundwater DOM, and gross primary production, standing crops of benthic organic matter, large woody debris and debris dams, and respiratory export (Figure 2.1). Export of particulate and dissolved organic matter was not considered for several reasons. Firstly, it is more strongly influenced by retention characteristics and hydrology than biological processes (Cummins *et al.* 1983, Webster *et al*. 1990). Secondly, spatial and temporal variation in organic matter retention and displacement (onto and from floodplains and sediment scour and fill) varies with the scale of flood events (Cummins *et al.* 1983). In addition the effect of a single large storm on organic matter transport may be very different between neighbouring streams (Webster *et al.* 1990). Capturing export during a range of events with widely different recurrence intervals (annual to 100+ years) requires a combination of long term data collection and very high collection frequency during high flow events. Cummins *et al.* (1983) recommend this occur at selected sites where existing long-term data sets can be continued and where improved understanding of storage (and export) dynamics can be obtained. Finally, without access to continuous monitoring equipment for flow data and continuous sampling equipment for transport data, annual estimates are likely to have wide errors. Webster

et al. (1990) showed that estimates of annual transport derived from continuous versus daily sampling can differ by a factor of three. Minshall (1996) suggested that for organic matter transport, sampling should occur during the rising and falling limbs of major changes in a stream's hydrograph.

A partial budget that focuses on inputs and storages of organic matter will resolve several key questions in stream ecosystem function. The relative importance of allochthonous and autochthonous resources, the relative magnitude and patterns of community metabolism and the size and distribution of organic matter storage can be determined. When partial budget data are collected for nearby reaches with similar disturbance history and climatic history, but different riparian vegetation the impact of riparian vegetation type on some fundamental ecosystem processes can be elucidated. As comparative biota studies in willow and native vegetation lined reaches have been inconclusive in both Australia and New Zealand, directly investigating ecological and physical processes may be fruitful. The present study can be considered complimentary to biota studies and necessary to build a complete picture of impacts from willow invasion at organisational levels from the individual to the ecosystem (Townsend and Simon 2001).

STUDY SITE DESCRIPTION

Criteria used to select study catchment

This chapter describes the study catchment and experimental reaches chosen for a comparison of organic matter dynamics within streams lined by willow and native riparian vegetation. Criteria were established to aid in the selection of sites that varied with respect to riparian vegetation but had low between site variability due to climatic, edaphic, historic or landuse factors.

A small order forested catchment was desirable as it is in such a catchment that litter fall from riparian forests is expected to be of greatest significance to stream organic matter dynamics (Vannote *et al.* 1980). Selecting a small catchment potentially limits unexpected hydraulic and geomorphic variation between reaches especially if the geology and landuse patterns are reasonably uniform.

In Victoria, willow recruitment has predominantly been by asexual propagules that have been carried downstream by streams and rivers (Ladson *et al.* 1997, Schulze and Walker 1997). A typical vegetation pattern has emerged in many catchments: native riparian vegetation upstream of willow dominated riparian vegetation. The change in riparian vegetation often coincides with land use changes along a catchment's geomorphic gradient. The steeper upper parts of many catchments along the Great Dividing Range in Victoria are dominated by forestry, water supply and conservation reserves and remain dominated by native vegetation. Lower in the same catchment agriculture may be the dominant land use. Observations in central Victoria were that the uppermost willow presence along many streams could be traced to either river management works or landscape plantings that coincided with land use change. An upstream-downstream study intending to compare the influence of willow and native riparian vegetation on stream ecology could be regarded as spurious when catchment land use and geomorphology differ markedly between sites (Barmuta 1999). As such, reaches with uniform catchment landuse and ones in which the riparian vegetation type (willow and native) alternated along a stream were seen as highly desirable. Paired reaches with similar catchment size, landuse, geology and geomorphology but different riparian vegetation type were also considered.

Disturbance was also seen as a crucial variable requiring control. The Telescoping Ecosystem Model (Fisher *et al.* 1998) suggests that time since disturbance determines the material processing characteristics of stream corridor ecosystems. Thus a study of material dynamics, in this case organic matter dynamics, that failed to address time since disturbance could again yield spurious results. Many native riparian forests can be considered remnants, that is they have not been cleared, burnt or grazed heavily enough to 'flip' them to an alternative ecological state (Petersen *et al.* 1998). Willow lined reaches were most commonly observed on extensively cleared landscapes where the previous native riparian vegetation has been removed. To avoid comparison of low disturbance native sites with high disturbance willow sites a common disturbance history was required. Alluvial mining and catchment clearing during the 1850's gold rush 'reset' succession within riparian forests alongside gold bearing streams. Consequently, these streams were considered desirable, especially when post gold rush land use had been similar adjacent to and upstream of willow and native lined reaches. Further discussion of gold mining history and consequences for riparian vegetation are found in section 3.4.

Study catchment location, geology and soils

A catchment was located that contained features closely conforming to the desired selection criteria. In addition accessibility was very good with a network of walking trails following 19th century water races. Groundwater sampling and understanding were facilitated by an extensive and well documented (Shugg 1996) development of mineral and fresh groundwater bores and spring discharges.

The regional setting, local geographic features of the study catchment and the location of the six study sites are indicated in Figure 3.1. The town of Daylesford, latitude 37°21' S., longitude 144°08' E., altitude 600 m and approximately 50 km north east of Ballarat, was situated in the geographic centre of the study catchment. Wombat Creek and Sailors Creek originated along the Great Dividing Range to the south of Daylesford and flows north. Wombat Creek flows into Sailors Creek to the west of Daylesford. Spring Creek originates to the north of Daylesford and flows north west to join Sailors Creek north of Hepburn Springs. The streams discharge into Jim Crow Creek, a tributary of the Loddon River. Classification of stream order was

complicated by alternating intermittent and perennial reaches along all streams. Using 1:25 000 map sheets and accepting the flow status indicated, stream orders were 1-5 using the Horton system as modified by Strahler (Knighton 1984).

The geology of the area is dominated by Palaeozoic interbedded shale and sandstone (Heislers 1993, Land Conservation Council 1985). Cainozoic volcanic activity had laid a thin veneer of basalt over approximately 25% of the study catchment. These formed as small lava fields or valley lava flows that now display inverted relief, with the resistant basalt occurring as a flat topped capping on ridges and small plains (Heislers 1993). The dominant volcanic region within the catchment extends as an approximately 2 km wide band from the southern limit of Sailors Creek to approximately 1 km north of a volcanic cone in the centre of Daylesford. Isolated volcanic ridge capping occurs within Hepburn Springs. The consequences of these geological sequences for groundwater hydrology are expanded in Chapter 5.

Soil types are different on the volcanic and sedimentary land units. Within the sedimentary geology interbedding of harder sandstone and softer shales occurs giving rise to a pattern of sandstone outcrops on ridges and, occasionally, steps on slopes. Soils in these locations are characterised by large amounts of stone (Mein and Bieniaszewska-Hunter 1987). The softer shales erode more easily yielding residual and transported clay minerals and occasional steep slopes. The resulting soils are yellow or, sometimes, red, mottled, acid and gradational or duplex. (Land Conservation Council 1985). Soil depth increases from ridges (<20 cm depth) to drainage lines (>100 cm, Mein and Bieniaszewska-Hunter 1987). Both clay deposits and coarse textured alluvium were observed along drainage lines. Sometimes extensive low mullock heaps were a conspicuous artefact from the gold mining era that could be considered a 'soil' type along drainage lines.

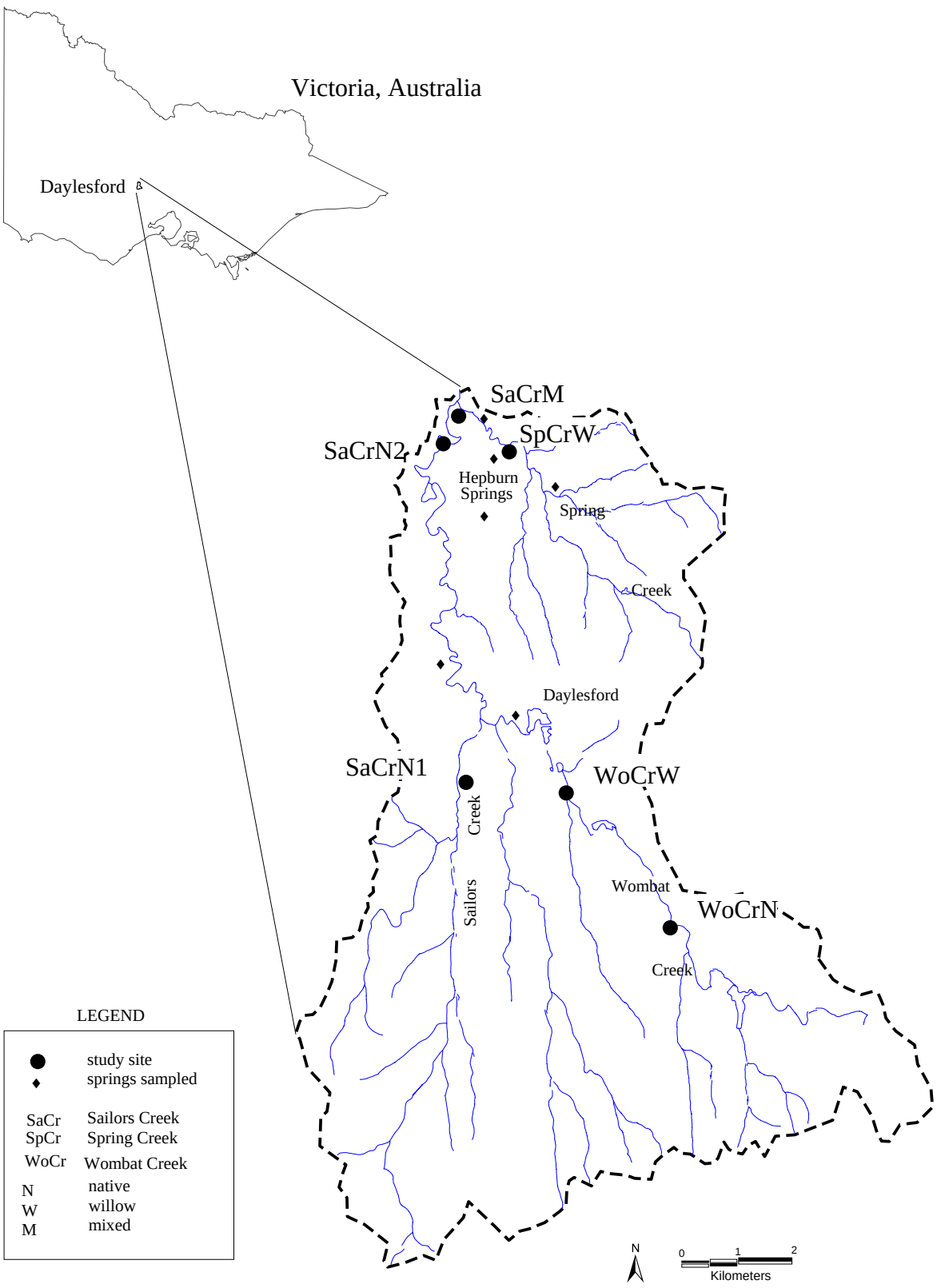


Figure 3.1. Study catchment location showing the six experimental reaches and groundwater sampling locations used in the present study. Map derived from Vicmap 1:25 000 digital maps.

Basalt from volcanic activity weathers to friable red gradational soils on the better drained upper slopes or brown gradational soils on lower slopes. Intensive agriculture in the catchment is restricted to these soils. In drainage lines the same basalt parent material weathers to black gradational cracking clays with slowly permeable subsoil (Land Conservation Council 1985) but these were localised and rare in the study catchment.

Climate

Bureau of Meteorology rainfall data have been collected at Daylesford (station 88020) since 1867 and relevant data were obtained directly from their records. Mean annual rainfall for Daylesford was 886 mm with an average of 129 rain days per year (Figure 3.2). The rainfall total was dominated by winter and spring fallswith, on average, 61% of the annual total falling in the period June to November. This dominance was reflected in raindays data. The mean number of raindays were 5-7 and 15-16 per month in summer and winter respectively.

There was a rainfall gradient across the catchment. Along the top of the Great Dividing Range, at the southern edge of the catchment, annual totals were approximately 1000 mm per annum. This decreased to approximately 800 mm along the northern edge of the catchment (Land Conservation Council 1985). The seasonal pattern of rainfall had clear consequences for the pattern of discharge from the catchment, with maximum flows and virtually all spates occurring between July and January (Figures 3.3, 3.11).

Temperature data have not been collected at Daylesford, but a 35 year record (1966-1999) exists for Castelmaine (station 88110, Figure 3.2), approximately 20 km north of Daylesford. Castelmaine is slightly lower in altitude (452 m) and drier (mean rainfall 613 mm). This is likely to be reflected in slightly higher mean temperatures. The ranges (i.e. mean daily minimum to mean daily maximum) were 11°-29°C for summer months and 2.6°-13°C for winter months. Temperature extremes recorded were -6.3°C in July and 43.7°C in January.

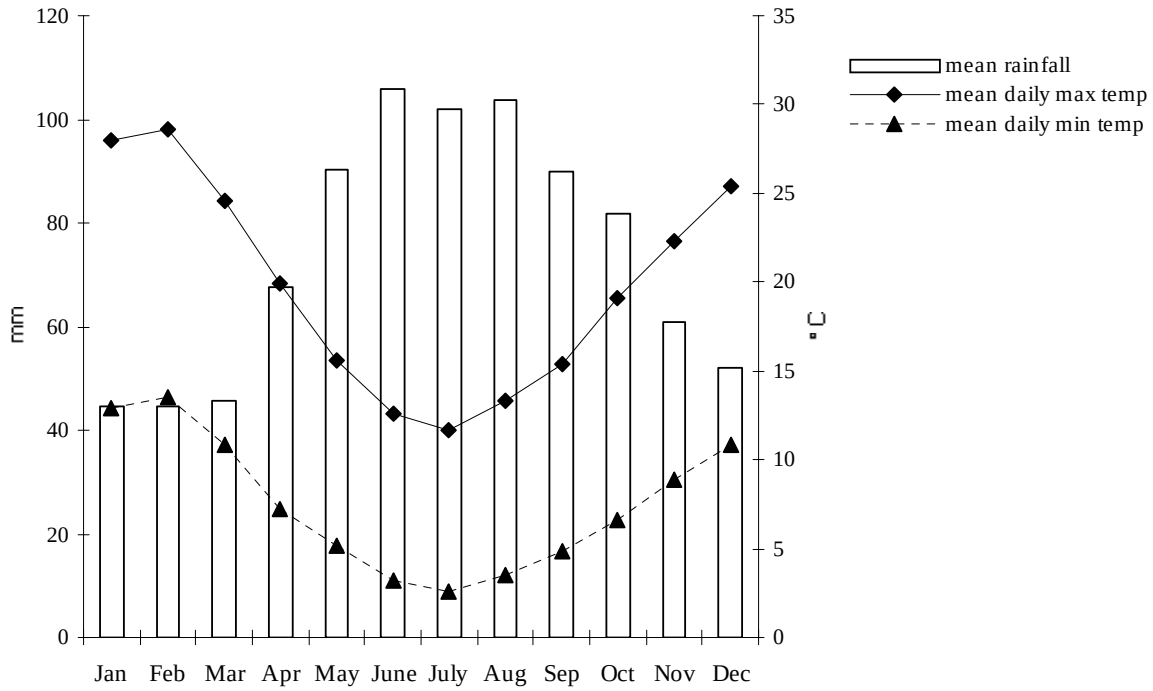


Figure 3.2. Mean monthly rainfall (left scale) for Daylesford (1867-1993) and mean daily maximum and mean daily minimum temperature (right scale) for Castelmaine (1966-1999). Data from Bureau of Meteorology.

During the present study, conditions were drier than average. Annual rainfall totals for Daylesford in 1997 – 1999 were 671, 777 and 814 mm respectively, compared to the long term annual average of 886 mm. For the 17 months between October 1996 and May 1998 monthly totals were below average for all but six months (Figure 3.3). Between 1996-97 and 1998-99, summer and early autumn mean daily maximum and minimum temperatures were above average. The drier and warmer than average conditions had implications for the persistence of surface stream flow in the study catchment (Section 3.6).

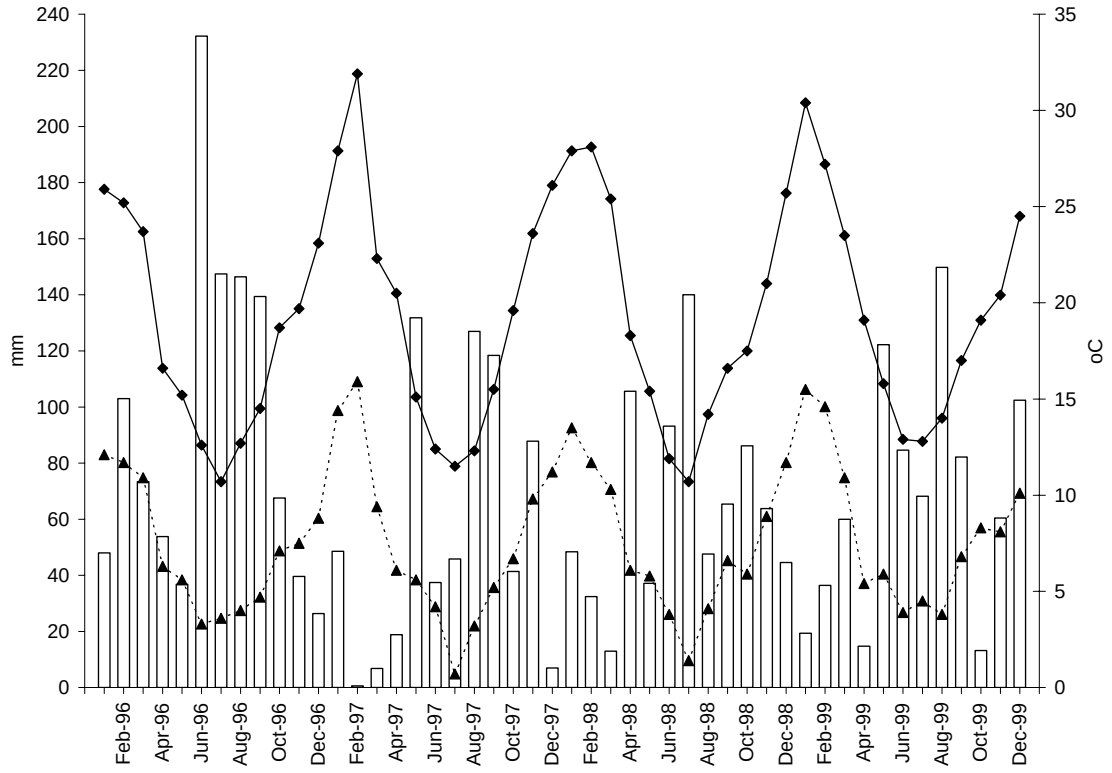


Figure 3.3. Mean daily maximum (solid line, right scale) and minimum (broken line, right scale) temperatures for Castlemaine and monthly total rainfall (columns, left scale) for Daylesford between January 1996 and December 1999, the duration of the present study. Data from Bureau of Meteorology.

Gold mining and landuse history

An 1869 review of the gold mining districts and modes of mining in Victoria (Smyth 1980, p107) described the Daylesford district in the following way:

> "The Hepburn goldfields are situate (sic) on the southern sources of the River Loddon. Jim Crow Creek - the main stream - and all its tributaries are auriferous, and many smaller creeks to the eastward support a numerous mining population....
>
> .... Sailor's Creek, Wombat, Spring Creek, Yandoit, Middleton Creek, Kangaroo Creek, and many smaller gullies have in times past been remarkable for great yields of gold from quite small areas, and there is scarcely one of them which does not even now renumerate the industrious miner....
>
> ....The old ground has been worked so often that vast quantities of sludge have accumulated in many places; and when sometime back a flood carried away

> much of the sludge and tailings, and opened new courses, the miners took advantage of the changes thus effected and worked the lower stratum, which was found to contain a great deal of gold."

Wombat, Sailors and Spring Creeks are the subject of this thesis and were all worked by alluvial miners. The pre-existing ecological conditions within streams and riparian zones and the transformation that occurred during the mining were not documented. A qualitative picture can be inferred from historical information including diaries, sketches and paintings. Eugene von Guerard's (1864) painting of alluvial mining in Spring Creek shows mobile sediments in the stream bed and a completely cleared and disturbed riparian zone. Goats are shown ranging the sparse vegetation of the valley sides. A similar picture is painted in words by Howitt (1857, p114), writing about the upper Yackandandah Creek in 1853:

> "We have begun to destroy the beauty of this creek. It will no longer run clear between its banks, covered with wattles and tea-trees, and amongst its shallow parts overgrown with foreign-looking shrubs, flags, and cypress-grass. A little while, and its whole course will exhibit nothing but nakedness, and heaps of gravel and mud. We diggers are horribly destructive of the picturesque.
>
> The creek runs about fifty yards to the left of our tent, and is, perhaps, some ten or a dozen feet across; but the stream only occupies part of this space, running amongst thickets of the afore-mentioned shrubs. Well, we set to work in earnest as soon as our tent was complete."

Approximately one month later he writes:

> "But what a change in so short a time! These valleys which, when we came up them, were so solitary, and so dense with tea-trees and wattles shrouding the courses of the stream, are now all studded with tents. The trees are felled by thousands; the creeks are laid open in long stretches to the day, by the tea-trees and scrub being cut down... These revolutions here are about as rapid as the shifting of scenes in a theatre."

Howitt was not writing directly of the Hepburn/Daylesford goldfields. However, he makes similar observations of Bendigo Creek (p254), Creswick's Creek and Spring

Creek at the Creswick diggings (p375), Campbell and Fryers Creeks (p363 and 364) and others. Combined with the observations of Smyth (1980) and von Guerard (1864) of the Hepburn Diggings an impression is created of immense deforestation and sediment mobilisation.

Following the gradual decline in alluvial mining in the late nineteenth century, eucalypt regeneration appears to have yielded a timber resource that has been exploited since the early part of the twentieth century. In many riparian zones cut stumps of 0.5 m or greater diameter were conspicuous. Logging, whilst continuing within the catchment, has been prohibited in the riparian zone since the 1989 Code of Forest Practice and from the 1950's loggers 'took additional care near streams too large to jump across' (pers. comm., DNRE forestry officers, Daylesford 1999).

Oral history from local families (D. McKinnon and V. Howell, pers. comm. Hepburn Springs, variously between 1996 and 1999) suggests frequent burning and constant grazing maintained the Spring Creek flats as treeless tussock grass dominated landscapes until post World War II. Their recollection is that families no longer maintained small grazing herds and house cows as affluence increased in the post war period. The grazing and burning pressure on individual reaches was likely to have been determined by the proximity to human settlement. Within the streams and riparian zones of the study catchment disturbance from mining, logging and domestic grazing have all been declining since the middle of the 20th century.

At the time of the present study the catchment was approximately two thirds native forest, contained within sections of the Wombat State Forest, Dry Diggings State Forest and all of Hepburn Regional Park. The remaining area was agricultural and urban (Daylesford and Hepburn Springs had a combined population of approximately 6000). Small areas of exotic conifers have been established in the Stewarts Creek Experimental Catchment within the Wombat Creek sub-catchment (Mein and Bieniaszewska-Hunter 1987) and as isolated plantations in other parts of the catchment. The combined area was believed to be less than 40 ha.

Another land use is water supply and impoundments. Wombat Creek Dam is a water supply reservoir of approximately 586 ML constructed in 1964 in the upper reaches of Wombat Creek. Hepburn Reservoir is approximately 45 ML and was constructed in 1921 in the upper reaches of Spring Creek. Water has not been extracted from Hepburn Reservoir since the early 1990's due to nuisance algal blooms. Jubilee Lake, Daylesford Lake and Slum Dam are ornamental lakes with no extraction licenses and all occur on the lower reaches of Wombat Creek.

Surprisingly few roads cross the streams and local residents perceived most riparian zones as difficult to access even when close. The exception being in cleared agricultural zones and within mineral spring reserves. These areas constitute approximately 15% of the total riparian length (Table 8.2). Amateur anglers and prospectors occasionally visit the streams. An indication of the low level of human impact was that on only one occasion was a piece of experimental equipment interfered with despite permanent installations for two years in six locations, some within 100 m of houses.

Vegetation and experimental reach selection

The dominant native vegetation of the region has been described by the Land Conservation Council (1985) as Open Forest II-III, dominated by *Eucalyptus obliqua, E. viminalis, E. radiata* and *E. dives*. Understorey is variously described as heathy, scrubby or layered with small trees dominated by *Acacia* species, particularly *A. dealbata*.

Riparian vegetation reflected a decreasing rainfall gradient from south to north. The upper reaches of Wombat and Sailors Creeks were dominated by *Eucalyptus ovata*, *E. viminalis* and *E. rubida* (Figures 3.4, 3.5, 3.6). Understorey tree and shrub species included *Acacia melanoxylon, A. dealbata* and *Pomaderris aspera* with *Poa labillardieri* dominating the ground layer and ferns and blackberry along the stream edges. Along drier more open gullies in the lower reaches of Sailors Creek *Leptospermum scoparium, L. lanigerum* and *Callistemon sieberi* sometimes dominated the shrub layer with fewer *Pomaderris* and ferns (Figure 3.7). Along the middle reaches of Wombat and Sailors Creeks and mid to lower reaches of Spring

Creek willows were a conspicuous riparian component (Figures 3.8, 3.9, 3.10). Taxonomy of willows in Australia is uncertain (Cremer 1995) but using Cremer (1995) and Ladson *et al.* (1997) the species were assigned to *Salix fragilis* var. *fragilis* and lesser numbers of a subspecies or hybrid within a *Salix cinerea* complex described in Ladson *et al*. (1997).

Riparian vegetation, be it native or exotic, represents post gold rush colonisation. All vegetation was viewed as late 19th and 20th century colonisation of mine sites with varied intervening disturbance regimes of fire, grazing and logging. The time of willow colonisation along Spring Creek has been estimated by Sniderman (1998) using dendrochronology and oral history. He found willows to be similarly aged, approximately 75% having established over a period 35-45 years previously (i.e. 1953-1963). This appeared to coincide with a decline in grazing and burning. Negligible willow recruitment had occurred in the 15 years prior to Sniderman's study and successional processes were observed with Sycamore and European Ash appearing capable of replacing willows. *Acacia melanoxylon* age classes, shade tolerance and distributions were not measured by Sniderman, but he suggested that they also appeared able to replace willows (J. Sniderman, pers. comm. University of Ballarat 1998).

Riparian vegetation 'communities' were defined and their occurrence within the study catchment surveyed. Experimental reaches were selected that were either lined by willows or native forest, with one site having a mixture of both (SaCrM). An alternating sequence of native-willow-native-mixed sites was achieved along a stream continuum from the headwaters of Wombat Creek to the lower reaches of Sailors Creek. Two sites were on side branches of this continuum; one native (SaCrN1) and one willow (SpCrW). SpCrW and WoCrN were paired sites with similar catchment size and land use and were used for community metabolism experiments (Chapter 6). Table 3.1 describes the vegetation at each site. Native sites were located in the wetter and drier vegetation communities described above and willow sites were located in forests dominated by both *Salix fragilis* var. *fragilis* and *Salix cinerea* (ssp. or hybrid).

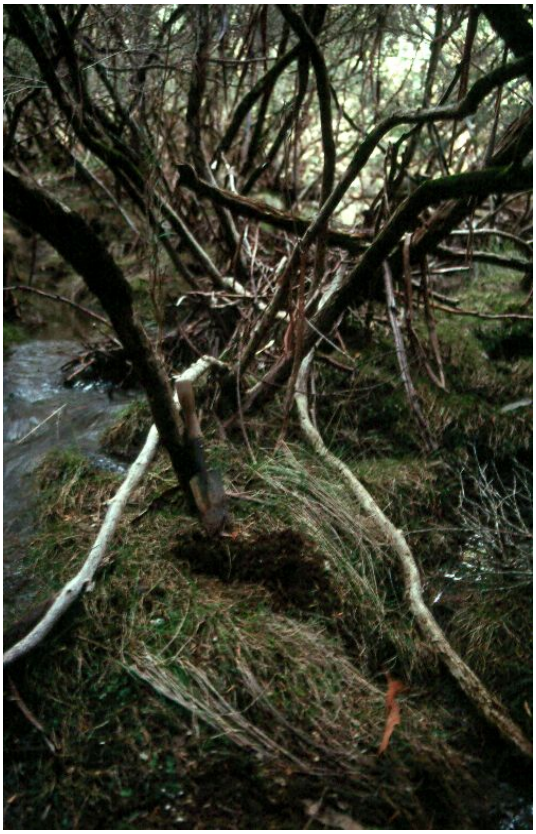
Figure 3.4. WoCrN with riparian community dominated by *Eucalyptus viminalis*, *Eucalyptus ovata*, *Acacia melanoxylon* and *Pomaderris aspera*. Groundcover was dominated by tussocks of *Poa labillardieri*. Photograph taken in early summer, 1m wide litter trap provides scale.

Figure 3.5. Sailors Creek upstream of SaCrN1 showing *Eucalyptus viminalis* and *Acacia melanoxylon* riparian canopy with the exotic *Rubus fruticosus* spp. agg. (Blackberry) the dominant riparian groundcover. Photograph taken in winter with surface stream approximately 3.5m wide

Figure 3.6. Detail of SaCrN1 showing riparian vegetation as in Figure 3.5 with the addition of *Cassinia aculeata*, *Pomaderris aspera* and *Poa labillardieri* in the understorey. Photograph taken in spring with 1m wide litter trap as scale

a

Figure 3.7. (a) SaCrN2 showing *Eucalyptus viminalis* dominated canopy and *Callistemon sieberi, Leptospermum lanigerum and L. scoparium* understorey. Photograph taken in winter with surface stream approximately three metres wide.



b

Figure 3.7. (b) Detail of SaCrN2 showing density of *Leptospermum* stems. Photograph taken in spring with bleached fallen branch in front left of photograph 1.2 metes long.

a

b

Figure 3.8. (a) Location of SaCrM showing a mixed riparian community of native and exotic species dominated by *Eucalyptus* spp., *Acacia dealbata*, *Acacia melanoxlyon* and *Salix fragilis* var. *fragilis*. Photograph taken in early summer. (b) Detail of SaCrM with *Salix fragilis* var. *fragilis* on left bank and *Eucalyptus vimialis* on right bank. Photograph taken in winter with width of surface stream seven metres.

a

b

Figure 3.9. (a) Location of SpCrW showing *Salix fragilis* var. *fragilis* dominated riparian vegetation with native forest (left hand side) and small plantation of *Pinus radiata* (right hand side) on adjacent slopes. Photograph taken in winter, house adjacent to plantation provides scale. (b) Detail of SpCrW site showing dense and low canopy of *Salix fragilis* var. *fragilis* and exotic herb ground cover. An example of the large woody debris (LWD) and associated debris dam typical of the site is visible in centre of photograph. Photograph taken in spring, well lit piece of LWD in centre of photograph was 0.41 m in diameter.

Figure 3.10. Detail of WoCrW showing dense and low canopy of *Salix fragilis* var. *fragilis* and *Salix cinerea* (Section 3.5 discusses *Salix* taxonomy) with trunk of *Acacia melanoxylon* visible to left of litter trap. Photograph taken in summer, 1 m wide litter trap as scale.

Stream physical and chemical characteristics

Catchment area, estimated stream bed gradient and mean daily discharge for each study site are summarised in Table 3.3. Mean annual discharge for each study site was estimated from the model derived by Nathan and Weinmann (1993). They provide an estimate of mean annual flow per square kilometre (163 ML km^{-2}) for the 166 km^2 catchment of the Jim Crow Creek gauging station at Yandoit. This catchment includes the study catchment. The mean daily discharge indicates the relative magnitude of flow between sites, but does not provide understanding of variability within a year. The daily discharge at Yandoit gauging station was used as an indication of the relative magnitude and frequency of spates within the study catchment (Figure 3.4). Spates occur almost exclusively between July and January with magnitude and frequency in dry years (e.g. 1997 and 1998) contrasting those of wet years (e.g. 1992 and 1996). As the study catchment occupies 56% of the gauging station's catchment it was assumed that spate frequency and relative magnitude would be comparable at the study sites. Absolute magnitude of individual spates would logically be lower.

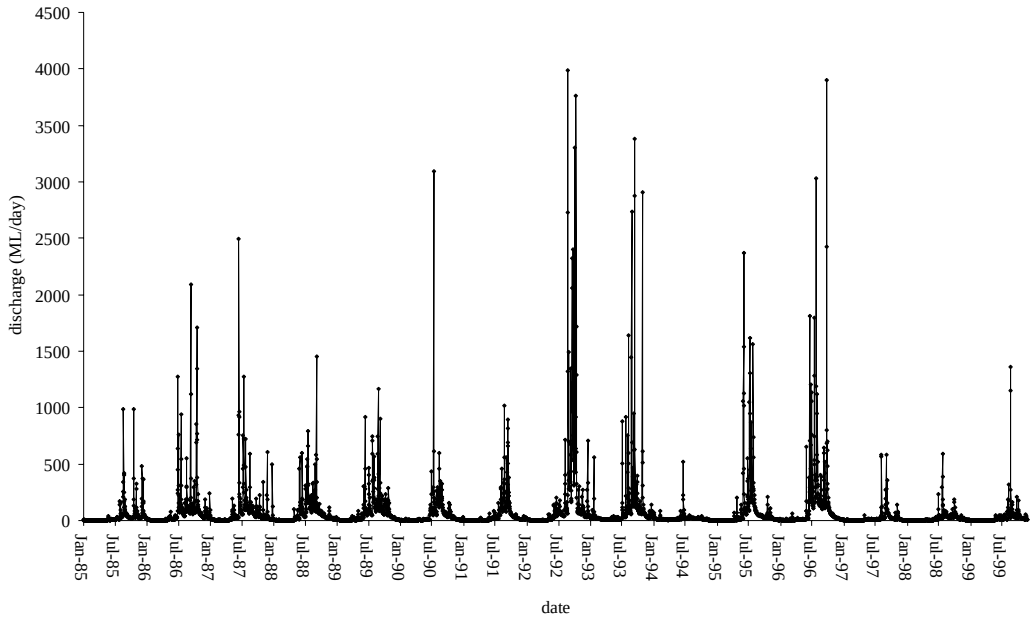


Figure 3.11. Mean daily discharge at Yandoit gauging station on Jim Crow Creek between January 1985 and November 1999. Data from Theiss Environmental Services gauging station 407221.

The present study was conducted during a period when spate magnitude was lower than all but one (1994) of the last fifteen years. These low flow conditions reflect below average rainfall totals (Fig. 3.3) and resulted in cessation of surface flow in some sections of Sailors Creek during late summer in 1998 and 1999. Local knowledge suggests that this is not an uncommon event even following wet winters, and it happens earlier when the spring and summer is dry (D. MacKinnon, V. and C. Howell, Hepburn Springs, pers. comm. 1998, 1999)

Stream bed area (2.3×10^5 m^2) was calculated from mean stream width multiplied by total stream length. Stream width was up to 7 m wide at larger order reaches (SaCrN2 and SaCrM) but averaged 2 m at other sites. Mean stream width for the catchment was estimated to be 2 m. Total stream length (116 km) was obtained using a length query within GIS software (MapInfo, MapInfo Corporation) for all stream segments shown in Figure 3.1 derived from VicMap 1:25 000 digital maps.

Site	Catchment area (km^2)	Order	Gradient (m/m)	Mean discharge (Ls^{-1})	Stream power (relative to SaCrM)	Vegetation - canopy - sub canopy trees and shrubs - groundcover
WoCrN	16.5	4	0.019	84	1.0	-*Eucalyptus ovata, E. radiata, E. viminalis* -*Acacia melanoxylon, Pomaderris aspera, A. dealbata* -ferns, *Poa* tussock
SpCrW	17.2	4	0.020	88	1.1	-**Salix fragilis, A. melanoxylon* (few) -**Crataegus monogyna* (few) -exotic herbs, *Poa* tussock
SaCrN1	22.1	4	0.017	113	1.2	-*E. viminalis, E. radiata* -*A. melanoxylon, P. aspera*, gorse -**Rubus fruticosus* spp. agg., native herbs
WoCrW	24.6	4	0.017	125	1.3	- *S. cinerea, S. fragilis, A. melanoxylon* (few) -**Crataegus monogyna* (few) -exotic and native herbs
SaCrN2	73.6	5	0.012	375	2.8	-*E. viminalis, E. radiata* -*Leptospermum spp., Callistemon sieberi, A. melanoxylon, A. dealbata* -*Poa* tussock, **Rubus fruticosus* spp. agg.
SaCrM	92.8	5	0.012	473	3.6	-*E. viminalis, E. radiata* -**S. fragilis, A. melanoxylon,* -*Poa* tussock, **Rubus fruticosus* spp. agg.

Table 3.1. Catchment area, stream bed gradient, estimated mean annual gradient and vegetation for six study sites on streams surrounding Daylesford, Victoria. Area, order and gradient data from 1:25 000 topographic maps, discharge data estimated from Nathan and Weinmann (1993), relative stream power from the product of discharge and slope, vegetation from the present study.

Conductivity, pH, dissolved oxygen and maximum and minimum temperature were recorded during community metabolism experiments (Chapter 6). The water authority responsible for Wombat Creek and Hepburn Reservoirs (Central Highlands Water) has collected data for these water bodies (Table 3.2). These provide characterisation of the water quality throughout the study catchment, whilst acknowledging that the potential nutrient loads from urban areas could have altered some parameters downstream of Daylesford and Hepburn Springs.

	Wombat Reservoir	Hepburn Reservoir
pH	7.1 (5.8-8.9, n=51, 1968-1992)	7.4 (6.7-9.2, n=34, 1968-1992)
Conductivity ($\mu S.cm^{-1}$)	71 (59-92, n=51, 1968-1992)	116 (56-160, n=34, 1968-1992)
Oxidisable nitrogen ($mg\ L^{-1}$)	0.2 (1994-1997)	0.8 (1984 and 1994, n=4)
Total phosphorus ($mg\ L^{-1}$)	0.013 (1994-1997)	0.03 (1984 and 1994, n=4)

Table 3.2. Chemical characteristics of water within Wombat and Hepburn Reservoirs. Means (range, n=number of samples, years data collected). Data from Kinhill Engineers (1994 Table 4.1) and Central Highlands Water Laboratories, pers. comm. Ballarat, 1999.

Spatial configuration

The ecosystem description used in the present study is adapted from Fisher *et al.* (1998). Organic matter dynamics were studied at a reach scale and total catchment dynamics derived from weighted means of reach data grouped by riparian vegetation type. The landscape unit of study was the stream-corridor ecosystem. Fisher *et al.* suggested this ecosystem consisted of four subsystems or components: the surface stream, hyporheic zone, parafluvial zone and riparian zone. In the present study distinction between the surface stream and the stream-corridor ecosystem was most important in quantifying groundwater DOM inputs (Chapter 5). Unless specifically stated organic matter budget parameters are quantified for the surface stream component of the stream-corridor ecosystem.

Study design consideration and constraints

Alternating native and willow sites along a stream continuum were selected to avoid the problem of upstream low disturbance native vegetation lined reaches being compared to downstream high disturbance willow lined reaches. Time since

disturbance and successional stage of riparian vegetation influence material retention, export and standing crop depletion within streams (Trotter 1990, Webster *et al.* 1990, Fisher *et al.* 1998). In the goldfields district of central Victoria widespread stream disturbance and catchment deforestation accompanied nineteenth century gold mining and early twentieth century logging and streamside grazing (Section 3.4). Streamside disturbance within forest reserves has been minor in the last 50 years since riparian harvesting controls were placed on logging operations and domestic grazing excluded. For the sites in the present study, willows established approximately 50 years ago. Based on these factors, time since major disturbance was considered to be similar at all sites.

It was assumed that reaches of similar order were hydrological similar as they were in close proximity, and drained catchments of similar size. There is a rainfall gradient across the study catchment from north to south (Section 3.3) that may lead to differential hydrological behaviour in the three main creek systems. Spring Creek, with the most northerly sub-catchment, may be expected to have a lower mean annual discharge per km^2 of catchment than Sailors or Wombat Creek. In addition, the timing and magnitude of storm events may differ, leading to differences in flood regime.

In organic matter budget terms, variation in hydrology has most impact on organic matter export (eg Webster *et al.* 1990), a factor deliberately not quantified in the present study (Section 2.5). Variation in hydrology does alter disturbance regimes, which have been suggested as a key influence on organic matter storage (Cummins *et al.* 1983, Webster *et al*. 1990, Fisher *et al.* 1998). A stream reach recently impacted by a 1 in 100 year flood event would be expected to have very different levels of organic matter in deep storages than an equivalent reach that had no recent history of large flooding (Cummins *et al.* 1983). In the present study, the effects of the gold rush appear to be a dominant factor in deep channel storage. The combination of mining activity and floods (Section 3.4) appears to have resulted in reaches that are either bedrock dominated with very coarse and shallow substrate depths or are invaded by riparian root mats (Chapter 7). In bedrock dominated channels, minor spates probably scour all organic matter except LWD, so even gross hydrological

differences probably have fewer consequences for channel organic matter storage than they would in channels with deep and soft substrates. Riparian root mats appear able to withstand exceptional spates. This was indicated by the widespread presence of stone walls dating from the gold rush period that had channel-side coverings of riparian root mats from *Salix*, *Leptospermum*, *Callistemon*, *Acacia melanoxylon* and *Pomaderris*. Floods over the last 150 years have not been able to dislodge the stone walls nor the root mats growing over them. It was thus assumed that the amount of organic matter stored in reaches of equivalent order between would not be strongly determined by hydrological variation.

A spread of native and willow sites within and between sub-catchments was seen as ideal, but opportunities for replication within catchments were limited by ecological history. Two general approaches have been taken in such circumstances. Firstly, paired catchments or reaches are frequently used in geomorphological and ecological theses (e.g. Pidgeon 1978, Treadwell 1995, Brooks 1999, Reed 1989, Read 2000). When only one individual of each 'treatment' is utilised the design potentially suffers from Hulbert's (1984) accusation of pseudoreplication. Pidgeon (1978) had one reach of each riparian vegetation type, Brooks (1999) had one disturbed and one undisturbed catchment and Treadwell (1995) had one reach of each stream order along a continuum This means inferential statistics (frequently ANOVA) used to test for differences between treatments may have been incorrectly applied. It does not mean that the descriptive work lacks merit. Brooks (1999) primarily used the paired catchments to build a model of what the disturbed (cleared and desnagged) catchment may have looked like prior to European settlement. and supported this with paleo-channel investigations within the disturbed catchment. Treadwell (1995) investigated a claim that the ratio of gross primary productivity to community respiration (P/R) altered along a river continuum. The aims of, and contributions from these works are informative and would remain so even if inferential statistics had been used as an analytical tool.

A second approach is to source replicates of a treatment from a wider geographic area. Read's (2000) willow-invertebrate study in Tasmania is an example. In this case, the increased replication of willow and native vegetation lined reaches

increased the conformity with inferential statistics as there was a fuller consideration of variation within and between treatments. However, there was a compromise, with a corresponding increase in variability of other potentially causal factors. For example, to achieve increased replication all willow sites were in disturbed reaches with cleared adjacent land and all were downstream of their paired native riparian vegetation reach in low disturbance forested catchments. Thus the inferential statistics could show differences between grouped treatments, but the ecological significance is marred by the inability to separate variation due to changes in riparian vegetation and variation due to land use change or relative position along a river continuum.

In the present study, three native vegetation lined stream reaches are compared to two willow lined reaches, with a reach lined by a mixture of natives and willows used to further explore patterns. Inferential statistics were used to compare data from the reaches and when 'treatments' (ie. riparian vegetation type lining the reach) were replicated. When 'treatments' were not replicated (e.g. community metabolism data in Chapter 6) inferential statistics could be used to investigate the probability of data sets coming from the same population, but any differences could not be attributed to differences in riparian vegetation type.

Unless variables can be manipulated such as the clearcut vs. unharvested design used in forestry impacts assessment (Webster *et al*. 1990), experimental designs in ecology are often due to chance events. In the present study, the catchment was selected to optimise the validity of between treatment comparisons but constraints were still present. Firstly, prior catchment land-use and disturbance, whilst of benefit to equalising disturbance history, could be seen as limiting the relevance of the study to other locations. Secondly, the spring fed nature of the streams may make them hydrologically dissimilar to other Victorian streams. One response to both these constraints is that these factors are uniform across all treatments thus the between treatment comparisons are valid. In addition, inverted landscapes that have resulted from valley lava flows (Section 5.1.1) are widespread in the Victorian Goldfields (Heislers 1993). As a result, spring fed valleys with a gold rush history are common.

A third constraint is the presence of reservoirs and ornamental lakes within the catchment. Reservoirs have been shown to influence hydrology, organic matter dynamics (particularly downstream export) and sediment dynamics (Ward and Stanford 1983, Walker 1985, Erskine *et al.* 1993). For example, when a low order stream (<3rd order) is dammed, the ratio of coarse particulate organic matter (CPOM) to fine particulate organic matter (FPOM) is hypothesised to decrease relative to an equivalent undammed reach (Ward and Stanford 1983). This is due to the higher retention rates of coarse organic matter relative to fine organic matter in a reservoir. In the present study the consequences of such disruption were equalised across treatments wherever possible with all but one site (SaCrN1) downstream of reservoirs (Figure 3.1).

A further constraint was determined by climate. In the period of study, rainfall was below average (Section 3.3) and surface flow ceased unusually early at some stream reaches (Section 3.6). Benthic organic matter alternated between submerged and exposed at some locations but remained submerged in the majority of mid channel sections of perennial, spring fed reaches. The two willow lined reaches and one native lined reach (WoCrW, SpCrW and WoCrN) had perennial surface flow across the majority of the reach whereas one native vegetation lined reach was completely dry on the surface (SaCrN1), and one native and the mixed vegetation site had only residual pools (SaCrN2 and SaCrM). It has been shown that exposed organic matter in temporary streams is slower to breakdown than equivalent submerged organic matter (Boulton 1991). This may have consequences for the amount of organic matter accumulated, with slower breakdown rates expected in temporary reaches and consequently greater accumulation for an equivalent accession. Flows were so low in late summer/early autumn that even in spring fed reaches flow was sub-surface in some sections and much organic matter stored in the channel was exposed. Comparison of benthic organic matter between reaches with different riparian vegetation type (Chapter 7) remained valid because the majority of benthic organic matter at all sites was exposed during at least some part of the year. In addition, within reach transects were used to explore the influence of riparian vegetation type on organic matter storage at the mixed site where there was uniform exposure of channel storages.

The temporary nature of stream reaches has a confounding influence on in-stream community metabolism. Aquatic autotrophic and heterotrophic activity is limited to residual pools or the hyporheic zone, with chemical conditions in the residual pools often stressful (Boulton and Suter 1986, Towns 1985). This is one reason community metabolism measurements were restricted to two sites (WoCrN and SpCrW) that remained perennial. A fuller justification for this choice is contained in Chapter 6.

LITTERFALL

Introduction

This is the first of four chapters to present and discuss experimental results for the study catchment described in the previous chapter. Litterfall is the subject of the first experimental chapter for two reasons. Firstly, the amounts, timing and composition of litterfall in evergreen and deciduous riparian vegetation is at the core of the hypothesis guiding this thesis. Secondly, litterfall has been shown to be the major organic matter input into forested upland streams overseas (Vannote *et al.* 1980, Webster and Meyer 1997a) though limited data suggest it may be less important in Australian upland streams (Pidgeon 1978, Treadwell *et al.* 1997).

The fall and subsequent decomposition of terrestrial litter represent a flux of energy and nutrients that has long been recognised (e.g. Ebermeyer 1876, cited in Bray and Gorham 1964). Terrestrial litterfall into aquatic systems represents a similar flux of energy and nutrients (Allan 1995). Quantification of the amounts and relative proportions of litter and other organic matter inputs, such as autochthonous primary production and groundwater dissolved organic matter (DOM), was pioneered by energetic studies at Bear Brook, USA (Fisher and Likens 1973). Since that study it has been widely argued, both theoretically (Vannote *et al.* 1980) and empirically (Webster and Meyer 1997a) that terrestrial litter is a major part of the energetic input into forested streams.

Lake (1995) reviewed published data on the quantity, timing and components of litterfall into streams and rivers in Australia. An emphasis was given to the seasonality of litterfall despite almost exclusively evergreen vegetation. In eucalypt forest and temperate rainforest peak rates of litterfall are in summer. Subsequent studies by Campbell and Fuchshuber (1994) in a cool temperate south eastern Australian forest, Greenway (1994) in a *Melaleuca quinquinerva* wetland and Clarke and Allaway (1996) in *Casuarina glauca* coastal wetland forests in eastern Australia have shown similar summer litterfall dominance. Where variation in the pattern of summer dominance has been observed it has been attributed to episodic events such

as storms and snow falls (Clarke and Allaway 1996, Campbell and Fuchshuber 1994) and branch or bark fall (Burrows and Burrows 1992, Greenway 1994).

A single unpublished study of exotic deciduous riparian litterfall in eastern Australia showed peak rates of litterfall in autumn (Pidgeon 1978). This is consistent with cool temperate northern hemisphere studies of litter accession into streams and rivers (Cushing *et al.* 1995). A contrast in seasonality of litterfall between native eucalypt dominated riparian vegetation and exotic deciduous riparian vegetation has been assumed in Australian studies, largely on the basis of Pidgeon's (1978) study or speculatively (Williams 1981, Campbell 1993, Schulze and Walker 1997). Campbell (1993) suggested that, based on data from Northern Hemisphere deciduous forests, the amounts of litter would be similar but the composition and seasonality would be noticeably different. If this was true, he argued, this would have substantial impacts on in-stream biota. Studies comparing biota under native and willow vegetation are inconclusive (Pidgeon 1978, Besley 1992, Hardwick *et al.* 1995, Schulze and Walker 1997) and with some authors suggesting other factors mask the effect of deciduous litterfall on the biota. For example, Schulze and Walker (1997) suggested that effects of willows along the River Murray may be so ubiquitous that invertebrate communities in reaches lined with River Red Gums and willows may no longer be distinguishable. The paucity of litterfall data for willows in the Southern Hemisphere appears to limit the understanding of their ecological impacts on Australian, NZ and South African aquatic ecosystems.

This chapter presents data for direct litterfall at the six study sites. Total litterfall and litter components are compared on an annual and seasonal basis within and between sites. Litterfall for the catchment as a whole and litterfall in relation to other sources of organic matter are discussed in the context of partial organic matter budgets prepared in Chapter 8.

Methods

Litterfall

Litterfall into streams was collected using 914 mm x 914 mm litter traps mounted horizontally on wooden stakes. Traps were of two types. Inverted rectangular prisms

of flexible PVC material were suspended from metal frames (Figure 4.1a). A netting (1 mm mesh) 'bag' was suspended from a 15 cm opening at the bottom and protected within a nursery pot that had drainage holes. These had been developed by the Centre for Forest Tree Technology (CFTT), Creswick and were borrowed for the present study. Additional traps were required so another was designed based on the CFTT traps. These had a netting (1 mm mesh) 'cylinder' hanging from an identical sized frame. When gathered at the base and tied with cord (Figure 4.1b) it formed an inverted rectangular prism of the same dimensions as the plastic traps.

4.1a 4.1b

Figure 4.1. The two types of litter trap used in the present study installed at SaCrM (4.1a) and WoCrW (4.1b). Each trap is 914 mm square.

Both traps ultimately collected litter in the same netting material, had identical frame size (i.e. the opening through which litter fell) and very similar depth and shape. The traps were assumed to collect litter with equal efficacy, but no field tests were conducted as it was felt the inherent spatial variability of natural litterfall (Gosz *et al.* 1972) would require large trap numbers of each design to be installed over very small

areas under a uniform canopy. Differences, if any, would relate to how material slides down the sides of the collection 'funnels' and its subsequent storage in the trap bases. Testing would require spatially uniform litterfall, consisting of litter components representative of all vegetation communities to be studied, under diverse environmental conditions (particularly wind and rain). It was expected that between site litterfall variation would be high (Gosz *et al.* 1972) and mask between trap variation which was expected to be small.

Where bedrock interfered with driving of stakes, traps were suspended by rope from overhanging branches. During the first year of collection, traps were positioned over the middle of the stream between 750 mm and 1m above water level. Flooding in late winter resulted in loss of data and trap damage. Traps were positioned on the stream bank immediately adjacent to the stream in the second year. This raised trap height by up to 750 mm relative to stream height. Campbell *et al.* (1992) found significantly higher litterfall collected in traps in the riparian zone than over the stream for a forested, upland stream in Victoria. In their study the channel was 2 m wide and the riparian traps were up to 10 m away from the middle of the stream. In the present study, the lateral displacement of the bank mounted traps was small with traps generally 2 m and a maximum of 4 m from the equivalent mid-channel position. The canopy architecture over the channel and bank locations appeared indistinguishable. In addition, litterfall data were compared between years, which may have revealed if significant differences in litterfall may have resulted from the change of position. Obviously, without controls on inter-annual variation in litterfall (ie. no replicate traps within the stream and along the bank in either or both years) this would only be circumstantial evidence.

At each site two traps were fixed in position and three were moved to new positions at each collection event (Wilm 1946, Campbell *et al.* 1992, Campbell and Fuchshuber 1994). The initial location was randomly generated for all five traps and thereafter for the three roving traps at each collection event. Reaches were subdivided into 5m intervals. This was an arbitrary choice that allowed a feasible number of permanent markers on the stream bank and ensured a spread of canopy conditions were sampled during any collection period even if all traps were adjacent to each other. When traps

were positioned on the bank (year 2) even location numbers were on the 'west' bank and odd location numbers were on the 'east' bank of each site. Meandering obviously meant the terms 'west' and 'east' banks were not strictly accurate descriptions. However, they represent the left bank and right bank of all streams when facing downstream.

Litter was collected at periods between 31 and 111 days, with the later an exceptional period that occurred between January and March 1997 at WoCrN whilst litter traps at all other sites were set up. At each collection event litter in traps was collected from all sites within as short a period as possible (between two and 14 days). Collection events were intended to coincide with seasons, with summer defined by convention as December, January and February and the other seasons following in groups of three months. Unfortunately, the collection event on or around the 1st of December 1997 was unavoidably delayed until early January. This meant that the litterfall totals for this collection event were an average of spring and summer. Asynchrony between collection and season also occurred at some sites during the summer and autumn 1997 during the set up of litter traps. Data for each collection event at each site were allocated to a seasonal category on the basis of the season in which most days litter collection had occurred, rather than the season in which the trap was emptied.

Litter was air dried, sorted into type and weighed. Types of litter were site dependent but included leaves, twigs, bark, capsules/fruits, flowers and a miscellaneous category that included unrecognisable fragments, arboreal mammal droppings, insects, and lichen. Wherever possible material was sorted by species or genera. This resulted in up to 10 categories per location. Rainfall would have leached soluble material from litter prior to collection (Jensen 1974, Conners and Naiman 1984). Leaching within streams can reduce initial dry weight of leaves by up to 25% (Webster and Benfield 1986). Intermittent leaching by rainfall would be expected to be less than in streams but fewer studies have considered this. Boulton (1991) estimated rainfall and dew leached 10% of the initial weight of *Eucalyptus* leaves when litter packs were positioned on exposed sections of stream banks. Iversen *et al.* 1982 suggested leaching may make up 6% of the weight of unleached leaves. No attempt was made in the present study to consider leaching within litter traps as weight loss was

assumed to be small and, as the present study is comparative, leaching was assumed to be the same at all sites.

Ash content was determined for each litter category. In nearly all cases this was achieved by pooling material from at least three sites and for a few seasons for each category. Litter of *Leptospermum*/*Callistemon* was only obtained from a single site (SaCrN2) but was pooled from various seasons. For highly seasonal categories, particularly flowers, only material from a single season was obtained but could be pooled from consecutive years.

Ash content was determined following combustion of known masses of each litter category at 500-550°C for a minimum of 2 hours and dry weights converted to ash free dry weight (AFDW).

Data analysis

Wilm (1946) was primarily concerned with increasing the efficiency of data collection for microclimatic or other factors which vary in both space and time. His method and analysis enables estimation of population means and variances from small numbers of monitoring stations. Where the number of monitoring stations becomes sufficiently large relative to the experimental site a simple randomised array of fixed stations should give satisfactory results. Wilm (1946) was addressing situations where the monitoring stations occupied "only an infinitesimal part of even a single acre of land", for example a rain gauge within a forest. Since the litter traps within the present study occupy approximately 2% of the surface area of each experimental reach it was suitable to utilise data unmodified by Wilm's methods. Campbell *et al.* (1992) and Campbell and Fuchshuber (1994) used such methods in recent litterfall studies in Australia.

The use of roving and stationary traps increases the number of sample units within the experimental site from a minimum of five per site, if all traps were fixed, to a theoretical maximum of 29 if three traps were moved on nine occasions. Resulting data include a larger component of variation due to spatial heterogeneity in the later case. Comparisons between sites or within sites between seasons are more

conservative when within site variation due to spatial heterogeneity is increased. As a consequence Wilm's experimental protocol was maintained but analysis of variance (ANOVA) was the statistical tool applied. ANOVA and Duncan's multiple range *post hoc* test with critical ranges (α=0.05) using Statistica (StatSoft Inc., USA) software were used for all analyses unless otherwise stated. Data were tested for homogeneity of variances with Levene's test and, on the few occasions assumptions were not met, data were log transformed and retested for homogeneity of variance prior to ANOVA testing. Where the above testing showed no significant differences power analysis of the experimental design was undertaken using GPOWER (Faul and Erdfelder 1992).

Means were compared within site and season but between years. For each site seasonal data from consecutive years could be legitimately pooled if there were no significant differences between the years. Pooled data were then compared within site but between seasons. Mean annual totals were determined from this pooled seasonal data, i.e. the sum of spring, summer, autumn and winter means. The above analysis was performed for total litter and for each litter category.

Results

Ash content

Litter components formed two discrete groups on the basis of ash content with the exception of *Leptospermum/Callistemon* fruit that had a mean within those of the lower ash content group but wide variance (Figure 4.2, $F_{17,35} = 8.42$, $P<0.001$). Seven litter components had ash contents between 5.7% and 7.4%: *Pomaderris* leaves, miscellaneous, Willow leaves, Willow flowers, Willow bark, arboreal mammal faeces and *Pomaderris* flowers in decreasing order (Figure 4.2). Ten litter components had ash contents between 2.1% and 3.6%: *Leptospermum/Callistemon* fruit, *Acacia dealbata* leaves, twigs/wood, *Eucalyptus* fruit, native (predominantly *Eucalyptus*) bark, *Acacia melanoxylon* leaves, *Leptospermum/Callistemon* leaves, *Acacia* flowers, *Acacia* fruit and *Eucalyptus* leaves in decreasing order. *Post hoc* testing showed *Leptospermum/Callistemon* fruit spanned the two groups but differences within groups were minor.

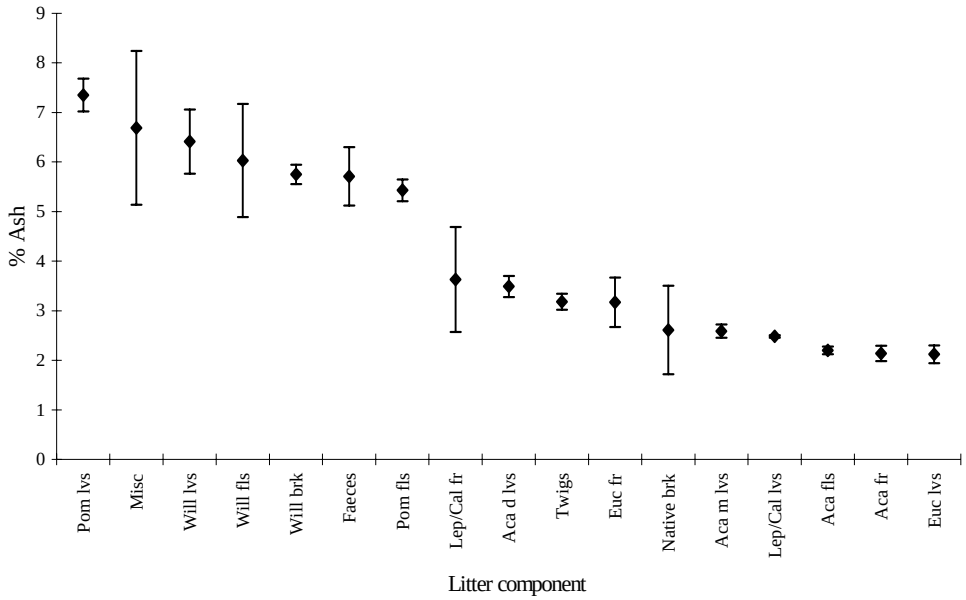


Figure 4.2. Ash content (% by weight) of 17 litter components collected in litter traps at six sites within the study catchment. Pom = *Pomaderris*, misc = miscellaneous, Will = willow, lvs = leaves, fls = flowers, brk = bark, Lep/Call = *Leptospermum/Callistemon*, Aca = *Acacia*, d = *dealbata*, Euc = *Eucalyptus*, m = *melanoxylon*. Means and standard errors of 2-6 replicates per component.

Total litter

Daily mean litterfall showed a strong seasonal pattern (Figure 4.3). Winter was the season of least litterfall at all sites and litterfall increased through spring to summer or autumn maxima. The period of peak litterfall appeared to be in early autumn at the willow sites and in summer at the native and mixed sites. The magnitude of the peak period of litterfall appeared slightly larger at the willow and mixed sites.

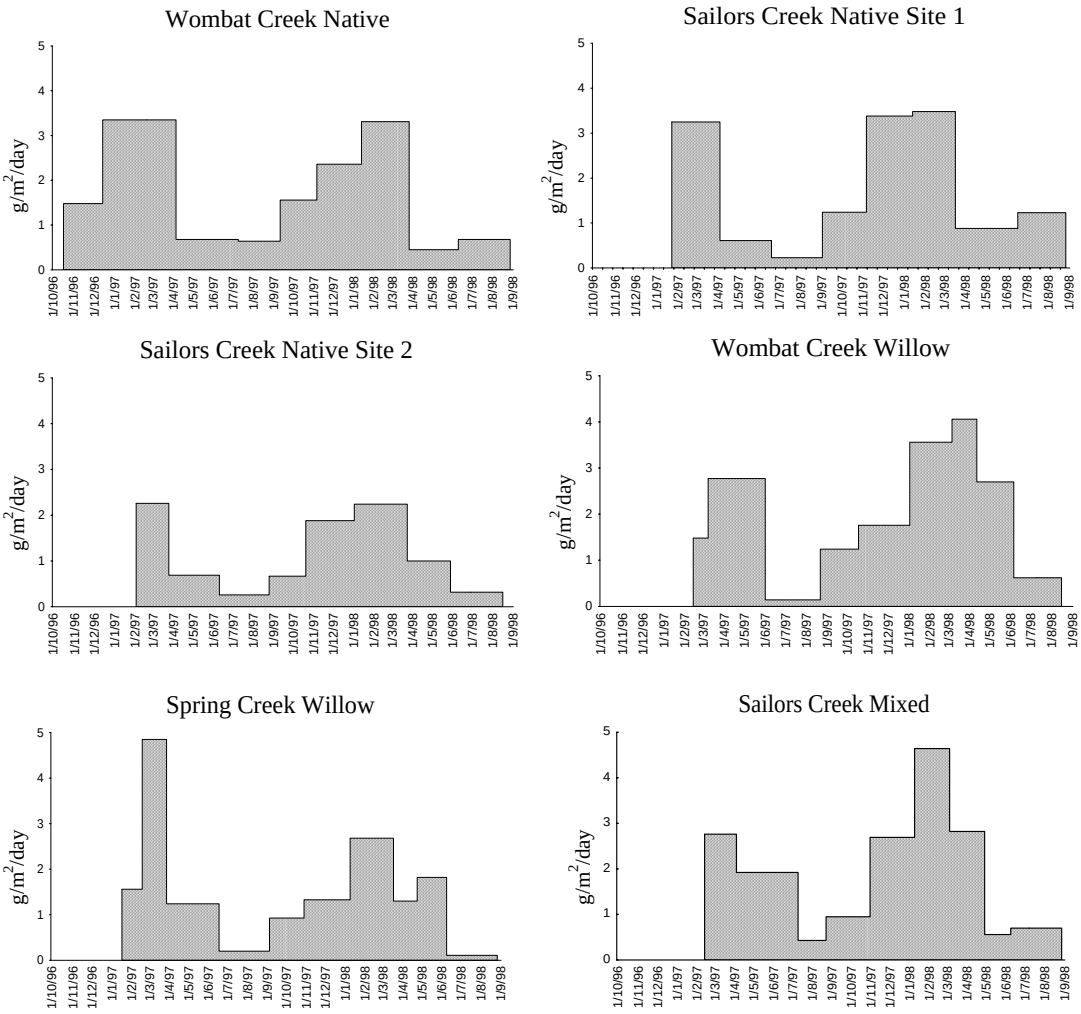


Figure 4.3. Daily mean dry weight of litter fall ($g/m^2/day$) into six stream reaches with differing riparian forests; three sites native forest, two sites willow dominated, one site an apparently even mix of natives and willows. Each step within the data represents a collection event, with 5 litter traps (area $0.84m^2$) per site.

Total litter data expressed as the mean daily ash free dry weight (AFDW) for each collection event were allocated to seasonal categories; spring, summer, autumn, winter and, where unavoidable, combined seasons (e.g. spring/summer). The seasonal data were then compared within sites between years. There were no significant differences between years for any season-site combination ($P = 0.16$, Duncan's *post hoc* test). Thus pooling of data from consecutive years was justified, yielding a single value for each season at each site (Table 4.1).

The validity of bulking daily litter accession data from native sites (WoCrN, SaCrN1 and SaCrN2) and willow sites (WoCrW and SpCrW) was tested by ANOVA within season between sites. For native sites there was a significant difference between at least two sites during at least one season ($F_{2, 119} = 6.18$, $P = 0.003$). Duncan's *post hoc* testing identified a significant difference between SaCrN1 and SaCrN2 in summer. This suggests bulking native site data was statistically justified for all but one of twelve combinations (i.e. three sites compared over four seasons). Data for native sites were thus bulked, whilst remaining aware of the one exception to statistical equivalence.

For willow sites there were no significant differences between the two sites in any season ($F_{1, 85} = 1.004$, $P = 0.32$) thus willow data were bulked. The mixed site was assumed to have characteristics of both vegetation communities and was analysed as a separate category.

	Daily litterfall (g m^{-2} d^{-1})								Total litter (g m^{-2} y^{-1})
	Spring		Summer		Autumn		Winter		
		%		%		%		%	
WoCrN	1.25	26	2.51	52	0.42	9	0.63	13	438
SaCrN1	1.22	20	3.29	54	0.54	9	1.03	17	555
SaCrN2	0.59	15	1.73	48	1.08	30	0.25	7	332
Mean native	1.02	20	2.51	51	0.68	16	0.64	12	442
WoCrW	0.99	17	1.93	34	2.44	42	0.39	7	526
SpCrW	0.92	20	1.83	39	1.81	39	0.13	3	427
Mean willow	0.96	19	1.88	37	2.13	41	0.26	5	477
SaCrM	0.77	15	2.61	49	1.42	27	0.52	10	486

Table 4.1. Estimates of seasonal daily litterfall (g AFDW m^{-2} d^{-1}), annual litterfall (g AFDW m^{-2} y^{-1}) and the seasonality of litterfall (as percent annual litter falling each season) for six stream reaches characterised by native (N), willow (W) or mixed (M) riparian vegetation communities.

Annual litter accession ranged from 332-555 g AFDW m^{-2} y^{-1}, with no significant differences between mean annual litterfall at native, willow or mixed sites ($F_{2, 3} = 0.11$, $P = 0.90$, power 0.1). Daily litterfall was significantly different between vegetation types in at least one season ($F_{6, 258} = 9.05$, $P < 0.001$). This was attributed to a significant difference between willow and native sites in autumn ($P < 0.001$, Duncan's *post hoc* test). No other significant differences were observed between vegetation types.

Within vegetation types there were seasonal differences in mean daily litterfall. At native sites ($F_{3, 127} = 51.18$, $P < 0.001$) summer was significantly different from all other seasons (all $P < 0.001$, Duncan's *post hoc* test). At willow sites ($F_{3, 89} = 13.38$, $P < 0.001$) both summer and autumn were significantly different from winter and spring (all $P < 0.02$, Duncan's *post hoc* test), but there was no difference between summer and autumn or winter and spring. At the mixed site ($F_{3, 42} = 7.51$, $P = 0.001$) summer was different to all other seasons and autumn was also different to winter (all $P < 0.03$, Duncan's *post hoc* test).

Seasonality of litterfall (expressed as percent of total litter falling each season) had a pattern that reflected the above observations of daily litterfall. Native sites had a highly seasonal pattern with approximately 50% of litter falling in summer and 12-20% falling over each of the other three seasons. SaCrN2 appeared to differ from the other native sites in having a larger proportion of litterfall in autumn. Willow sites were not dominated by a single season with approximately 40% of litter falling in summer, 40% in autumn, 19% in spring and 5% in winter.

Litter components

ANOVA with *post hoc* testing within sites and seasons and between years was undertaken seperately for each litter category. Sixteen of the 108 combinations were significantly different and nine of these were winter year one versus winter year two comparisons. Spates in the first winter had damaged many traps and some sites had data from only one trap. When winter comparisons were removed only seven (6% of the possible combinations) were significantly different with these spread over four of the six categories of litter and four of the six sites. As a result it was felt that combining consecutive years data was justified.

Leaves and twigs dominated nearly all sites in all seasons (Figure 4.4). Leaves constituted between 41-58% of the annual total at all sites. Leaves were least important as a percentage of total litter in winter (10-21%) at all sites except SaCrN2 (46% of total, significantly different from all other sites $P = 0.009$, Duncan's *post hoc* test).

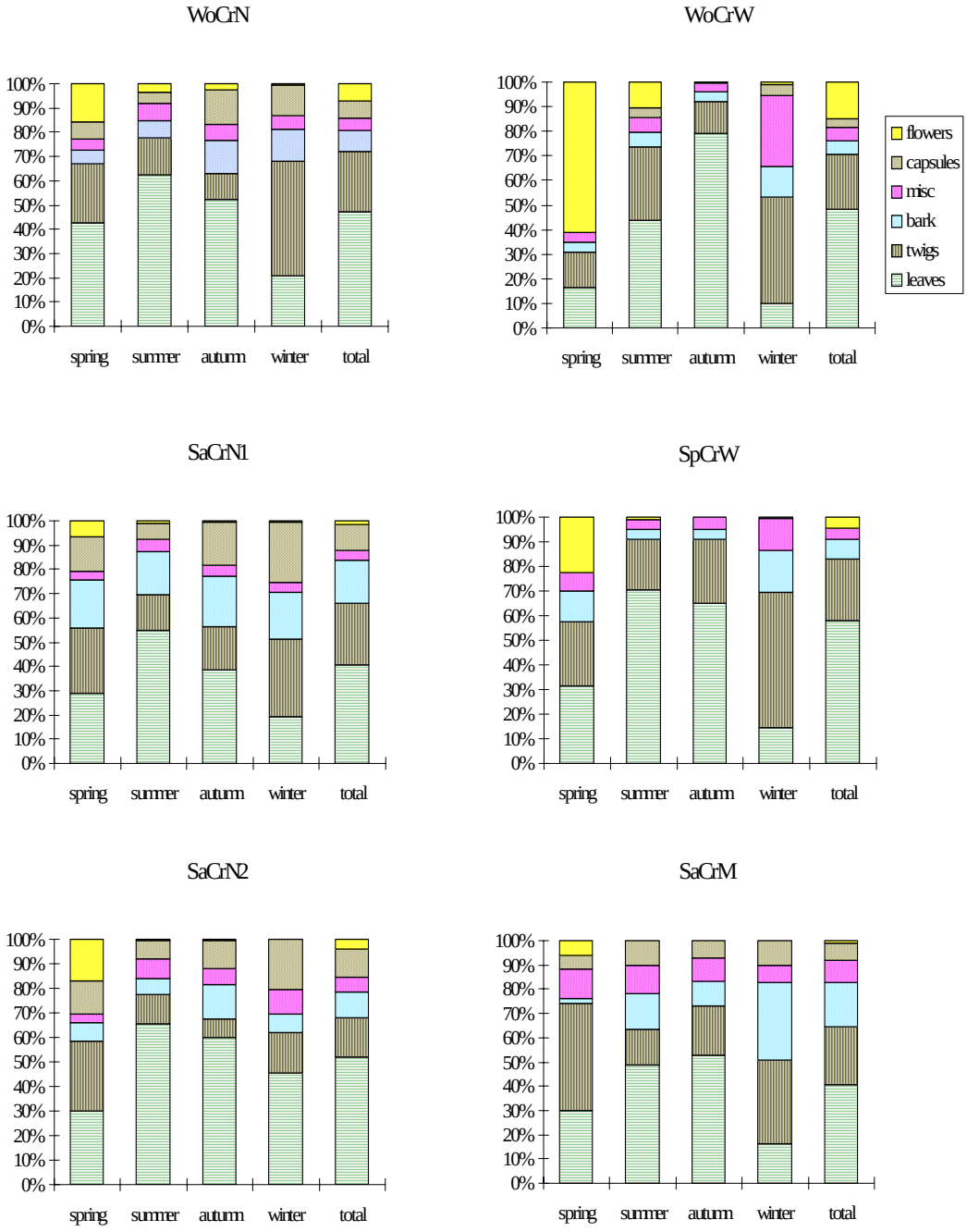


Figure 4.4. Percentage contribution of each of 6 litter categories (leaves, twigs, bark, capsules, miscellaneous and flowers) for each season and annually (total) for the six study sites. Three native sites (WoCrN, SaCrN1, SaCrN2) on the left and two willow (WoCrW, SpCrW) and one mixed site (SaCrM) on the right.

Twigs were the next largest component of annual totals at all sites (16-25%). Twigs dominated winter litterfall (32-47%) at all sites with the exception of SaCrN2 (16%, significantly different to all other sites, all $P < 0.05$ except SaCrN1, $P = 0.09$, Duncan's *post hoc* test). Twigs and leaves constituted 50 - 90% of the litter at all sites in all seasons except at WoCrW in spring which was dominated by flowers.

Bark rarely constituted more than 20% of litterfall at any site in any season with the exception of SaCrM in winter (39% of litterfall). There were no significant differences between vegetation types in the contribution of bark to total litterfall. (native sites, 8-17%; willow, 6-8% and mixed site, 19%), though this may be a result of low statistical power (0.1) rather than a lack of biological differences. It was observed that long strands of bark were shed by *Eucalyptus viminalis* and *E. ovata* and often fell close to the trunk of trees or remained hanging in the canopy. Willows and *Acacia melanoxylon* were similar in shedding smaller pieces of bark that did not hang in the canopy and were far less noticeable.

Flowers could be an important component of litter in spring (6-61% of litter) but were rarely important for annual totals (1-15%). *Salix cinerea* was a prolific catkin producer at WoCrW in spring and early summer (WoCrW was significantly different to all other sites in spring, all $P < 0.0001$, and to all sites except WoCrN in summer, all other $P < 0.04$, Duncan's *post hoc* testing).

Capsules (*Acacia*, *Leptospermum* and *Callistemon*) and miscellaneous litter were a consistent but minor contributor to all sites in all seasons (combined range, 4-32%). The lowest and highest values were from willow sites.

Discussion

Ash content

There was a clear distinction in ash content between willow litter and that of sclerophyllous native species. A non-sclerophyllous native species (*Pomaderris*) also had a high ash content in both leaves and flowers. The distinctions between taxa were more obvious than those between litter types with the exceptions that twigs and

woody fruits/capsules were typically low in ash and the miscellaneous and faeces categories high.

Nitrogen and carbon compounds are lost during combustion at the temperature used for ashing in the present study (Sutcliffe and Baker 1981). The main constituent of plant ash is potassium, often comprising 50% of the total weight of ash. Ash is typically rich in calcium, sodium, magnesium and phosphorous with most other minerals present in trace amounts (Sutcliffe and Baker 1981). Consequently, ash is a crude indicator of mineral content and has been used as a preliminary guide to the value of fodder in animal husbandry (Mackenzie 1970). Willows are highly palatable and a recognised shrub and tree fodder species, comparing well with lucerne hay as a ruminant feed (Reid and Wilson 1986). They are also palatable to aquatic invertebrates but feeding trials comparing preferences between *Eucalyptus* and willow leaves are inconclusive (Yeates 1994, Schulze and Walker 1997). Ash contents suggest a difference in mineral quality between willow and native sclerophyllous litter that may have consequences for aquatic food chains. To date Australasian studies have considered willow leaf breakdown rates, nitrogen and phenol content, and invertebrate feeding preferences (Pidgeon and Cairns 1981, Yeates 1994, Lester *et al.* 1994a, Schulze and Walker 1997). However, mineral budgets could be developed to explore the consequences of ash rich, palatable litter from willows for major nutrients, including nitrogen, phosphorus, calcium and magnesium in aquatic ecosystems (e.g. Gosz *et al.* 1972).

Litterfall

Rates of annual litterfall into the study reaches (332-555 g AFDW m^{-2} y^{-1}) were within the range documented for upland streams flowing through evergreen or deciduous forests in south eastern Australia (310-699 g m^{-2} y^{-1} Campbell 1993), South Africa (268-500 g m^{-2} y^{-1} Davies *et al.* 1995), New Zealand (282-429 g m^{-2} y^{-1} Winterbourn 1995), USA (median 448 range 202-963 g m^{-2} y^{-1} n= 19 Webster *et al.* 1995, median 528 range 313-730 g m^{-2} y^{-1} n=18 order <6 Benfield 1997) and Europe (466 g m^{-2} y^{-1} Dawson 1976, 655 g m^{-2} y^{-1} Iversen *et al.* 1982, 700 g m^{-2} y^{-1} Marxsen *et al.* 1997). Litterfall from forests generally decreases with latitude, precipitation and altitude, (Bray and Gorham 1964, Campbell 1993, Benfield 1997), but Bray and

Gorham (1964) gave averages of 550 g DW m^{-2} y^{-1} for warm temperate forests and 350 g DW m^{-2} y^{-1} for cool temperate forests. Annual litterfall in forests appears to be related to climate (Bray and Gorham 1964, Benfield 1997) far more than phenology (e.g. deciduous versus evergreen, Bray and Gorham 1964). This is consistent with the results from the present study where little difference was observed in annual litterfall from willow and native riparian vegetation.

Two litter size classes, one small and one large, were poorly sampled using litter traps. Very fine litter (<1mm) could pass through the netting used to collect litter. *Eucalyptus* stamens and *Acacia dealbata* pinnules were obvious litter components able to pass through the netting. These were occasionally numerous but their contribution to the total weight of litter was likely to be very small. Small litter traps as used in the present study are not suitable for measuring large woody debris (Bray and Gorham 1964, Gosz *et al.* 1972, Conners and Naiman 1984). A few stream reaches were surveyed separately for large woody debris standing crop (Chapter 7). Long strands of bark, especially from *Eucalyptus*, and branches sometimes fell across litter traps. The portions within the traps were retained and the portion suspended outside the traps discarded. It was assumed the traps were collecting twigs, small branches and bark effectively (Conners and Naiman 1984).

The timing of litterfall was linked to phenology in the present study. Native evergreen riparian vegetation generally exhibited a strong seasonal pattern, with 50% of litter falling in summer (Table . Willow riparian vegetation was less dominated by a single season, with 40% of annual litterfall in both summer and autumn. Campbell (1993) suggested that streams in south eastern Australia receive a smaller proportion of the annual litter accession in the season of peak litterfall and a larger proportion in the season of least litterfall than streams in northern hemisphere deciduous forests, i.e. they are less seasonal. The willow dominated vegetation in the present study did not follow the extremely seasonal pattern of northern hemisphere deciduous forests outlined by Campbell (1993). The seasonal pattern at willow sites was similar to that observed in an Australian cool temperate *Nothofagus* forest (Turnbull and Madden 1983) and wet sclerophyll forests (Ashton 1975).

Campbell (1993) suggested that another potentially significant factor was the proportion of annual litter falling in the season of least litter accession. In Australian native forests this season can be winter or autumn and the proportion of annual litterfall between 8-17% (from Campbell *et al.* 1992), values which are very similar to those of the present study. For northern hemisphere deciduous streams Campbell (1993) suggested that the season of least litterfall (winter) contributes only 5% of the annual total. In the present study willow sites had a similar winter contribution. The consequences of this relative difference in the season of least litterfall for stream ecosystems is unknown. Organisms dependent on direct consumption of fresh litter (eg shredders) are likely to be most affected. However, as least litter falls in winter when low temperatures may limit invertebrate productivity the net effect is difficult to predict. Studies of macroinvertebrate response to manipulated seasonal diets would be informative.

Non-leaf litter contributed a significant amount to annual litter fall (42-59%) at all sites in the present study, a recognised characteristic of Australasian warm temperate forests (Bray and Gorham 1964, Campbell *et al.* 1992). The willow and native sites had similar proportions of non-leaf litter. This is consistent with Webster *et al.*'s (1995) estimate of 42% non-leaf litter contribution to total litterfall into streams in the Eastern Deciduous Forest Biome of USA, but is contrary to the observations of Bray and Gorham (1964) for deciduous forests in the Northern Hemisphere (23% non-leaf litter). Twigs dominated the non-leaf litter at all sites with smaller and variable proportions of bark, capsules, flowers and miscellaneous materials.

In the Northern Hemisphere, *Salix* is typically a pioneer and the forest canopy is dominated by other genera (e.g. *Quercus*, *Fagus*, *Acer*, *Juglans*, *Platanus*, Fisher 1995, Webster *et al.* 1995). In the present study willows were the dominant deciduous plants and the only canopy species for many kilometres of stream edge. Litterfall patterns may be expected to be different in pioneer forests and later successional stages (Webster *et al.* 1990). *Salix fragilis* is well known for its fragile twigs, especially in winter (Ladson *et al.* 1997) and this is consistent with the dominance of twigs in winter litterfall at the willow sites in the present study. Other Northern Hemisphere deciduous genera may be less brittle leading to lower amounts of twig

fall for an equivalent canopy cover. These factors may explain why the patterns observed in the present study do not precisely mimic those expected from an analysis of Northern Hemisphere litterfall studies (eg Campbell 1993).

Transfer of patterns observed under deciduous canopies in the Northern Hemisphere to willow lined streams in Australia can be erroneous. Speculation based on such comparisons can clearly be useful in forming research hypotheses and this was the overriding objective (Bunn *et al.* 1993) of Campbell's (1993) review of riparian-stream linkages. The fact that the present study has found fewer differences than suggested may, in part, explain the lack of significant differences observed in some studies comparing invertebrates in willow and native lined reaches (Besley 1992, Hardwick *et al.* 1995, Schulze and Walker 1997).

A confounding variable in the seasonal patterns observed in the present study is floristic diversity. Native sites were diverse and the willow sites were floristically simple. WoCrW had a greater diversity than SpCrW with *Salix cinerea* (~60% of the canopy), *S. fragilis* (~30% of the canopy) and *Acacia melanoxylon* (~10% of the canopy) represented in the canopy, compared to *S. fragilis* and an individual *A. melanoxylon* tree in the canopy of the latter site. The mixed site was even more diverse with *Eucalyptus* as an overstorey, *A. melanoxylon* and *S. fragilis* as a mid-canopy and *Callistemon* as a sparse understorey. Lowest winter contribution is observed in the simplest willow site (SpCrW) and it increases with the floristic diversity of arboreal species. Thus the pattern observed reflected the site selection criteria, in which willow sites were chosen for their simplicity (i.e. they were as monospecific as possible) and native sites were chosen for their community structure (i.e. they included examples of a range of native riparian communities observed in the catchment). The mixed site appears to have a seasonal pattern between that of the native and the willow sites, reflecting both floristic complexity and the combination of deciduous and evergreen elements. However, the later influence is not a factor at SaCrN2 (all species are evergreen) yet its seasonal pattern is almost identical to the mixed site.

The only other comparison of riparian evergreen and deciduous forest litterfall in Australia was conducted by Pidgeon (1978) along a stream on the New England Tableland of NSW. The region had been heavily disturbed with extensive clearing for grazing. The riparian communities were very simple with few arboreal species other than exotic willows and native *Eucalyptus blakelyi*. Pidgeon compared litter accession into pools lined by an open eucalypt woodland, cleared pasture and willows. Unfortunately, the willows closely lined both banks of the pool but the woodland lined only one bank with a wide floodplain separating trees from the other bank by 35 m. Total litterfall was 57 and 457 g AFDW m^{-2} y^{-1} at the eucalypt and willow sites respectively. The proportion of litterfall in the season of maximum and minimum litter accession was 48% (spring) and 12% (autumn) at the eucalypt site and 62% (autumn) and 5% (winter) at the willow site.

The low annual litterfall at Pidgeon's (1978) eucalypt site reflected the floristic simplicity and sparse cover of the disturbed open woodland. The annual total and seasonal pattern at the willow site is similar to that observed in the present study. The seasonality was more pronounced reflecting a ten fold increase in leaf fall in autumn relative to summer. In the present study leaf fall in autumn was no more than three times that of summer. This may reflect the differences in the timing of litter collection events (Pidgeon, monthly; the present study, irregular intervals between 31 and 85 days at willow sites). The poor synchrony between litter collection, calendar seasons and leaf abscission in the present study resulted in summer litter samples containing recently abscised willow leaves. Even though these had been falling for a short period they may elevate the summer daily litterfall average. This appears to have been significant for SpCrW, where summer data in 1998 included the first 17 days of March (i.e. 18% of calendar autumn). In addition a dry summer and autumn gave an early beginning to leaf fall in willows. A similar issue arose for autumn data with the inclusion of some winter litterfall potentially lowering the autumn daily average.

Different species composition and possibly wetter conditions at WoCrW meant the collection interval was synchronous with leaf fall. Summer collections had few abscised willow leaves and a proportional increase in leaves in autumn was observed (Fig 4.3). However, the proportion of annual litterfall in each season was similar to

SpCrW, indicating that the timing of litter collection is not wholly responsible for the differences in seasonality observed by Pidgeon (1978) and the present study. Differences in climate, floristic diversity and species behaviour (Pidgeon studied *Salix cinerea* and *S. purpurea*, whereas willows considered in the present study were a *S. cinerea* hybrid and *S. fragilis* var. *fragilis*) could all contribute to different seasonal patterns.

Amounts, seasonality, composition and ash content were highly variable between seasons, between species and between and within sites. Such variability is a feature of litterfall studies generally (Gosz *et al.* 1972) and it limits confidence in patterns observed from small sample sizes. The data presented in this chapter are from only the second study of litterfall under exotic deciduous riparian vegetation in Australia. Autumn dominance is intuitively expected as one observes the willow canopy change colour and a carpet of bright leaves covering the ground. However, the indistinct fall of green leaves, twigs, bark and catkins in other seasons were very important in the present study and eliminated autumn dominance. If this pattern is widespread in Australia it may explain why hypothesised impacts of willow spread on invertebrate communities (Campbell 1993) have not been observed in some studies.

This chapter has considered the first of three stream organic matter inputs to be considered in the present study. The results from this chapter will contribute to a partial organic matter budget for the study catchment presented in Chapter 8. The next chapter discusses the input of dissolved organic matter in groundwater entering the streams.

GROUNDWATER DISSOLVED ORGANIC MATTER

Introduction

Groundwater contains dissolved organic matter (DOM) leached from decomposing organic matter in soils and from organic materials within the aquifer itself (Thurman 1985). Where groundwater contributes to stream flow and contains DOM it can be considered a source of organic matter to the stream corridor ecosystem. The input of DOM to streams has in the past been measured by recording flow and DOM content at known spring eyes throughout a watershed (e.g. McDowell and Likens 1988). Alternatively, an estimate of groundwater contribution to stream flow can be made from hydrographs and other hydrological techniques and the groundwater DOM content determined from piezometer or bore samples (e.g. Wallis 1979, Treadwell *et al.* 1997). In practice dissolved organic carbon (DOC) is measured and organic matter assumed to contain 45-50% carbon (Boulton and Brock 1999).

DOM in groundwater ranges from 0.4 to 30 mg L^{-1}, with a median value of 0.7 mg L^{-1} with the exception of trona water from oil shale regions (up to 80 000 mg L^{-1}) and oil-field brines (up to 2000 mg L^{-1}, Thurman 1985). There has been only one published Australian study that has estimated groundwater DOM inputs to a stream corridor ecosystem. Treadwell *et al.* (1997) determined shallow groundwater DOM content adjacent to Keppel Creek in central Victoria. A bore 1 m deep and 1 m from the stream contained 16.8 mg DOM L^{-1}. They assumed all stream flow originated as groundwater and calculated an annual DOM input of 5507 g AFDW m^{-2} y^{-1}.

Hydrogeology of the Wombat-Sailors-Spring Creek catchment

Shugg and Knight (1994) and Shugg (1996) reviewed the hydrogeology of the region and they described two aquifer systems; the Quaternary Basalt Aquifer (QBA) and the Ordovician Bedrock Aquifer (OBA). The former is characterised by fresh water and the later by both fresh and mineral water. The Quaternary Basalt Aquifer (QBA) is contained within solidified basaltic lava flows and associated volcanic materials

including tuffs and agglomerates (Shugg 1996). The lava flows form a discontinuous veneer (<60 m thick) overlying Ordovician sedimentary rocks. Springs frequently occur along the volcanic - sedimentary boundary. During the 1850's gold rush, miners followed paleo-streams buried by the lava flows (Smyth 1980). Many of these shallow small tunnels have collapsed but remain an effective conduit for QBA groundwater. Hence a large number of springs emerge at tunnel entrances. QBA water around Daylesford is relatively fresh being typically low in total dissolved salts (average approximately 200 mg TDS L^{-1}, Heislers 1993).

The OBA is contained within interbedded slates, shales and sandstones. The OBA is a fractured bedrock aquifer that Shugg (1996) suggests has two regions. The shallow groundwater flow system is characterised by relatively fresh water close to the Great Dividing Range (including the Daylesford region). The deep circulating system is characterised by mineral waters that have been observed at depths greater than 450m within mine shafts (Shugg 1996). Mineral spring eyes have been identified by Europeans since the 1850's gold rush and are well mapped and frequently have pipes, bores or pumps that allow collection by the public (Wishart and Wishart 1990). Bores within the OBA have been drilled for domestic purposes and allowed some access to the shallow circulating OBA fresh water.

Knowledge of the flow path from an aquifer to the stream is important when determining the DOM contribution to the stream corridor ecosystem. Groundwater from aquifers that deliver directly to the stream undergoes little or no transformation of its DOM constituents. Where groundwater is delivered to a component of the stream corridor ecosystem other than the surface water, for example the hyporheic or riparian zones, then there is opportunity for DOM constituents to be transformed (Fiebig 1995, Fraser and Williams 1998).

In this chapter the DOM content of the groundwater discharging and within the aquifers that supply the Wombat-Sailors-Spring Creek catchment are discussed. Changes in DOM content as a result of groundwater subsurface flow through riparian

sediments are discussed. In addition, the groundwater DOM contribution to both the stream corridor ecosystem and the surface stream component are assessed. This analysis will contribute to a partial stream organic matter budget for the Wombat-Sailors-Spring Creek watershed presented in Chapter 8.

Methods

Groundwater sampling

Dissolved organic matter sampling was conducted between spring 1997 and spring 1998. A total of 7 mineral springs, two QBA fresh water springs and one OBA fresh water bore were sampled for DOM content throughout the study watershed (Figure 3.1). Mineral springs with permanently flowing pipes rather than hand pumps were selected to reduce possible errors from inadequate extraction of bore volumes. Spring flow was measured for QBA and mineral springs. Shugg (1996) provides flow data for some mineral springs and provides an estimate of the total flow from Hepburn Mineral Springs Reserve (0.105 ML d^{-1}).

Twenty millilitre glass collection bottles and an accompanying small square of aluminium foil (approximately 30 mm x 30 mm) were purged of organic carbon by combusting at 500°-550°C for a minimum of two hours. The small square of foil was placed over the mouth of each bottle prior to screwing on the plastic lid thus acting as a carbon free membrane. Bottles remained closed until filled in the field and the foil was replaced prior to screwing on lids.

Water was filtered in the field using 0.45μm polypropylene filter paper (Gelman GH Polypro). A hand held, squeeze action vacuum pump, buchner funnel and evacuation flask from a Hach soil saturation extract kit allowed useful volumes of even slightly turbid water to be filtered quickly. Filter paper and acid washed filter apparatus were rinsed three times with ultra pure water (Milli-Q Plus). Water to be sampled was used to rinse the complete apparatus three times. A sample was filtered, collected and immediately acidified to pH<2 with two or three drops of concentrated hydrochloric acid. Samples were sent overnight to the Water Studies Centre, Monash University,

Melbourne for total non-purgable carbon analysis. Limit of detection was 1 mg L^{-1}. It was assumed that total non-purgable carbon was equivalent to organic carbon. Organic matter was assumed to contain 50% carbon (Thurman 1985, Boulton and Brock 1999) so laboratory results were multiplied by two to give organic matter content.

Replicate samples of mineral water were collected with and without filtration from 4 mineral springs in September 1998. Samples were acidified and analysed as above. It was assumed that significantly larger values for unfiltered samples would reveal the presence of particulate organic matter larger than 0.45 μm.

Demineralised or ultra pure water samples were used as a control. Replicates were poured directly into bottles, acidified and DOC determined as for groundwater samples. These controls tested contamination from rinse water and from acidification. Demineralised or ultra pure water samples were also passed through the filter paper and apparatus before collection, acidification and laboratory analysis. These samples tested contamination from the filter paper and filter apparatus.

Riparian zone transect.

The streams typically had a narrow flat floodplain within a bedrock defined valley. Springs occasionally discharged on slopes above the floodplain. As the spring water reached the flat floodplain water percolated through the riparian sediments as well as meandering slowly across the floodplain toward the stream. To determine DOM contributed to the stream from spring water flowing through riparian zone sediments, a transect was established in January 1997 from where spring water ponded on the floodplain surface to Spring Creek. The spring (QBA spring 1) emerges from an abandoned mine shaft approximately 80 m east along a walking track beginning at the end of Seventeenth Street Hepburn Springs. The transect was perpendicular to the stream flow and 11 m in length. Six shallow wells were dug to bedrock with a hand auger (100 mm diameter) along the transect.

Saturated hydraulic conductivity was determined at each well using the auger hole method. Water was bailed from the well and the subsequent rate of rise in water level recorded. Methods and nomographs followed Landon (1984). At each well the height of both the ground surface and the ground water table relative to the stream surface level was surveyed using a horizontal string line. A cross section of the riparian zone was thus constructed, showing the ground surface, water table, bedrock profile and stream bank.

Samples for dissolved organic carbon analysis were taken from QBA Spring 1 water ponded on the riparian sediments, each well and Spring Creek. Filtration, acidification and laboratory analysis were as described in section 5.2.1. Wells were bailed dry and allowed to refill three times immediately prior to sampling in an attempt to lessen possible diluting or concentrating effects within the well.

Groundwater DOM contribution

Two estimates of groundwater DOM contribution were obtained. The first was an estimate of DOM delivered to the stream corridor ecosystem based on the mean DOM content of groundwater measured within bores or at spring eyes multiplied by base flow. Nathan and Weinmann (1993) provided a base flow index (0.35) for the Yandoit gauging station on Jim Crow Creek. The study catchment occupies 56% of the Yandoit gauging station catchment. The mean annual discharge for the catchment was estimated in Chapter 3. Mean annual discharge multiplied by the base flow index gives an estimate of groundwater flow to the streams in the study catchment.

The second method estimated groundwater DOM contribution to the surface stream component of the stream corridor ecosystem. Groundwater either flowed directly into streams or flowed through riparian sediments prior to discharging into streams. Mineral OBA springs discharged or were piped directly into the stream. No shallow OBA seeps or springs were identified in the catchment and nearly all streams flow over Ordovician sedimentary geology. It was thus assumed that all OBA discharge (both mineral and fresh) delivered DOM directly to the streams. Basalt within the

catchment is typically ridge capping with frequent springs observable at the basalt/Ordovician boundary, thus QBA springs delivered water to the surface away from streams. These spring waters typically flowed as surface water down steep ridges before disappearing into riparian sediments on the narrow but flat floodplains. It was thus assumed that all QBA water was transformed as in the riparian transect described in section 5.2.2. The portion of base flow attributed to each aquifer (QBA and combined OBAs) was estimated from the proportion of each geology in the catchment. This was estimated from Land Conservation Council (1985) geological maps.

Results were expressed in g AFDW for comparability with results from other chapters and DOM content was assumed to equal AFDW content.

Data analysis

Contaminated sample data (section 5.4.1) were discarded prior to analysis. Filtered and unfiltered rinse water and mineral water sample means were compared using t-tests for independant samples (Statistica, StatSoft Inc). Mineral and QBA fresh water sample means were compared between seasons using analysis of variance (ANOVA, Statistica).

Results

Controls

There were no significant differences in DOC content of unfiltered and filtered demineralised (Table 5.1, t=0.85, 5 degrees of freedom, P=0.44) or ultra pure water (t=0.61, 10 degrees of freedom, P=0.55). September 1998 testing yielded two samples of filtered ultra pure water with organic carbon contents (38 and 12 mg L^{-1}) well above those of the third filtered replicate (<1 mg L^{-1}) and all other filtered and unfiltered samples (ranged from <1 to 3 mg L^{-1}).

Date	Un-filtered DOC (mg L^{-1})	Filtered DOC (mg L^{-1})
Nov 1997 (demineralised)	2 (0) n=3	2 (0.5) n=4
Feb 1998 (ultra-pure)	<1 n=1	1 (0.3)* n=3
May 1998 (ultra-pure)	1 n=1	1 (0.4) n=5
Sept 1998 (ultra-pure)	1 n=1	17 (19)* n=3
all ultra pure samples	0.8 (0.3)* n=3	1 (0.4) n=9#

Table 5.1. Mean and standard deviation (in parentheses) of dissolved organic carbon (DOC) content of demineralised or ultra pure water with or without filtration through 0.45 μm polypropylene filter paper mounted in field vacuum extraction apparatus. Organic carbon contamination by filter paper or apparatus would be indicated by significant differences in values for unfiltered and filtered water. n = number of samples. * indicates a value below level of detection (<1 mg DOC L^{-1}) was determined by the laboratory for one sample of a treatment cohort and an arbitrary value of 0.5 mg DOC L^{-1} was assigned to determine mean values. # indicates two outliers of filtered samples excluded (Section 5.4.1).

Groundwater DOM

DOM content of springs and one bore from the three main aquifer systems of the Daylesford region averaged approximately 4 mg L^{-1} (Table 5.2). There were no differences in DOM content of fresh water from the QBA in spring and autumn. However, mineral water from the OBA showed significantly lower DOM content in autumn relative to spring ($F_{2,18}$=23, P<0.001). The mean value for mineral water was 4 mg L^{-1}.

Date	DOM (mg L^{-1})		
	QBA, fresh	OBA, mineral	OBA, fresh
Nov 1997	4 (2) n=4	5 (1) n=6	
May 1998	4 (0) n=4	2 (0) n=8	4 n=1
Sept 1998		5 (2) n=7	

Table 5.2. Mean and standard deviation (in parentheses) dissolved organic matter (DOM) content of fresh water springs from the Quaternary basalt aquifer (QBA), mineral springs from the Ordovician bedrock aquifer (OBA) and fresh bore water from the Ordovician bedrock aquifer (OBA) at various times of the year. n = number of samples.

Particulate organic matter

There were no differences in DOM content of unfiltered and filtered mineral water (Table 5.3). No particulate organic matter larger than 0.45μm was detected in mineral water. A single unfiltered sample from Central Spring had a DOM content of 42 mg L^{-1}. The other replicate had a DOM content of 4 mg L^{-1}, similar to the mean of filtered samples (5.7 mg L^{-1}). Thus the sample with high DOM content was regarded as an outlier, with contamination suspected.

Spring	Unfiltered (mg L^{-1})	Filtered (mg L^{-1})
Central	4 n=1#	7 (1) n=2
Locarno	6 (3) n=2	4 (0) n=2
Tipperary	5 (1) n=2	6 (0) n=2
Wagga	6 (0) n=2	4 n=1
All	5.4 (1.5) n=7	5.4 (1.5) n=7

Table 5.3. Mean and standard deviation (in parentheses) of dissolved organic matter (DOM) content of mineral water with and without filtration through 0.45 μm polypropylene filter in September 1998. Unfiltered samples were collected directly from permanently piped spring eyes, acidified and immediately capped. For filtered samples mineral water flowed directly into filter apparatus from spring pipes and samples were immediately filtered, acidified and capped. #excludes a single outlier of 42 mg DOM L^{-1} for a Central spring sample (Section 5.3.3).

Riparian zone transect

In January 1997 QBA Spring 1 water that had ponded on the Spring Creek floodplain was 720 mm above the stream surface level and 11 m to the south. The water table was assumed to be at the ground surface at this point and it sloped at a relatively even gradient toward the stream (Figure 5.1). Approximately 4 m from the stream the water table assumed the same height as the stream surface and remained horizontal between that point and the stream. The bedrock profile had a shape similar to the water table. Thus the depth of water above bedrock remained relatively constant for

five wells (a range of 21.4 - 23.9 cm), with the remaining well having relatively less water (10 cm). The ground surface was slightly undulating.

DOM content of the riparian groundwater showed an increasing trend along the transect from surface spring water toward the stream bank (Table 5.4). Water from QBA Spring 1 appeared to flow from its "eye" at a collapsed mine tunnel entrance onto the riparian floodplain with no change in DOM content (4 mg L^{-1}). Ground water sampled from wells 9, 6.9 and 5.25 m from the stream, that intersected the sloping water table all had a DOM content of 8 mg L^{-1}. The DOM content of the ground water then increased to 10 mg L^{-1} within the wells 3.5 m and 2 m from the stream and to 14 mg L^{-1} within the well 1 m from the stream. Spring Creek had a DOM content of 8 mg L^{-1}.

Saturated hydraulic conductivity at the well sites showed a similar trend to that of the DOM content of the groundwater, increasing with proximity to the stream. Between 9 and 5.25 m from the stream saturated hydraulic conductivity was between 0.1 and 0.4 m d^{-1}. At 2 m from the stream saturated hydraulic conductivity was an order of magnitude greater (2.2 m d^{-1}) and 1 m from the stream it was nearly two orders of magnitude greater (9.9 m d^{-1}). The increase in saturated hydraulic conductivity coincided with increasing presence of riparian roots in auger spoil.

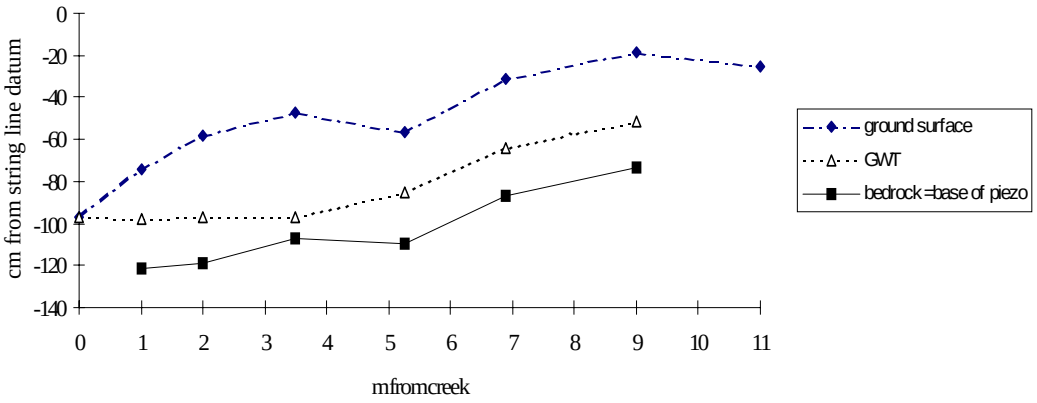


Figure 5.1. Schematic cross section of an 11 m riparian transect adjacent to Spring Creek showing ground surface, ground water table and bedrock profiles in relation to a horizontal string line datum (0 on Y axis). Data points represent piezometer locations except for 0 m (Spring Creek bank) and 11 m (surface ponded spring water).

dist from stream (m)	0*	1	2	3.5	5.25	6.9	9	11*
DOM (mg L^{-1})	8*	14	10	10	8	8	8	4*
Ksat (m d^{-1})	-	9.9	2.2	nd	0.1	0.15	0.4	-
GWT (cm)	-	-1	0.1	-0.1	12	33	46	72
land surface (cm)	-	23	39	50	41	66	79	72

Table 5.4. Dissolved organic matter (DOM) content of sampled water, saturated hydraulic conductivity (Ksat) and height of ground water table (GWT) and land surface relative to stream water surface for 6 shallow bores and spring and stream surface water (*) across an 11 m riparian transect adjacent to Spring Creek in January 1998. Shallow bores were augured to bedrock and sampled two weeks later (28/1/1998). nd = not determined.

Groundwater DOM contribution to the stream corridor ecosystem

Nathan and Weinmann (1993) calculated a base flow index of 0.35 for Jim Crow Creek at Yandoit, a catchment that includes the study catchment. This was used to estimate the base flow for the study catchment (0.35 x 1.5 x 10^4 ML y^{-1} (mean annual flow, Table 3.1) = 5294 ML y^{-1}). Base flow multiplied by mean groundwater DOM

content (4 mg L^{-1}, Table 5.2) gives the DOM contribution from groundwater to the stream corridor ecosystem (Table 5.5).

Groundwater DOM contribution to surface streams

The catchment's surface geology consisted of 23% Quarternary basalt and 77% Ordovician sedimentary. It was assumed that base flow was derived uniformly across the catchment and that surface geology reflected aquifer distribution. Consequently, 23% of base flow was assumed to be derived from the QBA (1218 ML y^{-1}) and 77% from the OBA (4076 ML y^{-1}). OBA groundwater was assumed to reach the surface streams without transformation so delivered 4 mg L^{-1} (=1.6 x 10^7 g DOM y^{-1}, Table 5.5).

QBA springs appeared to deliver water to the surface stream component predominately via flow through riparian sediments. For QBA Spring 1 an estimated 75% of the flow to the surface stream was subsurface through the riparian zone and DOM content increased to 14 mg L^{-1} as a consequence (Table 5.4). All other QBA springs observed in the catchment appeared similar with a steady flow down steep ridges that infiltrated the riparian sediments on reaching narrow flat floodplains. It was thus assumed that 75% of the QBA discharge delivered 14 mg DOM L^{-1} and the remaining 25% delivered 4 mg DOM L^{-1} to the surface stream component of the stream corridor ecosystem (=1.4 x 10^7 g DOM y^{-1}, Table 5.5). OBA and QBA contributions were summed to give an estimate for the surface waters within the catchment (Table 5.5).

Groundwater DOM contribution can be expressed on an area basis by dividing annual contribution by stream bed area (Table 5.5). Total stream length was 116 km with an average width of 2 m and a stream bed area of 2.3 x 10^5 m^2.

	GW discharge (ML y^{-1})		DOM contribution (g AFDW y^{-1})	(g AFDW m^{-2} y^{-1})
stream corridor ecosystem	5294		$2.1x10^7$	91
surface stream component	OBA	4076	$1.6x10^7$	70
	QBA	1218	$1.4x10^7$	61
	total	5294	$3.0x10^7$	137

Table 5.5. Groundwater (GW) discharge (mean annual discharge in ML y^{-1}) and groundwater DOM contribution expressed as a total (g AFDW y^{-1}) and on an area basis (g AFDW m^{-2} y^{-1}) for the stream corridor ecosystem and the surface steam component of that ecosystem within the study catchment. DOM content was assumed to equal AFDW of organic matter.

Discussion

Assessment of method

Field filtration and use of polypropylene filter membranes appeared to result in insignificant organic carbon contamination of samples. This is illustrated by the similarity of filtered and unfiltered samples of demineralised, ultra-pure and mineral water (Tables 5.1 and 5.3). Where differences occurred they were dramatic, with DOC content of samples an order of magnitude greater than other replicates. This only occurred three times during 49 sampling events and could have been caused by a single or small number of relatively large organic matter particles contaminating a sample. Replication, use of new filter membranes for each sample and rinsing with both ultra pure water and the water to be sampled effectively reduced contamination to rare but easily identified occurrences. The three contaminated samples were clear outliers and were discarded in analysis of data.

Groundwater DOM

Groundwater DOM content of three main aquifer systems within the Daylesford region appeared to be similar, averaging 4 mg L^{-1} (Table 5.2). The relatively shallow QBA system appeared to have no seasonal differences in DOM content, whereas a distinct seasonal difference was observed in mineral water (Table 5.2). Shugg and Knight (1994) used hydraulic characteristics of Ordovician bedrock and Darcy's Law to estimate a minimum travel time of 1 to 100 years for mineral water from the

recharge sites to the Hepburn Mineral Springs Reserve. Despite the very wide range of the estimate, such travel times would appear to preclude the observed seasonal (ie. < 0.5 year) pattern in DOM. Whilst the cause of the seasonal fluctuation in DOM was not directly investigated it may be a result of seasonal differences in mixing ratios with shallow groundwater or ambient surface waters rather than an intrinsic characteristic of the deeply circulating mineral water.

Shugg (1996) describes a mixing zone where ascending mineral water mixes with shallow groundwater and/or ambient surface water prior to discharge. Quantity and solute concentrations of shallow groundwater and surface waters have been shown to be closely related to seasonal rainfall patterns (e.g. Fraser and Williams 1998). In autumn, following characteristically dry summers, shallow and ambient water would exert lower potentiometric pressure (low flows were observed in streams and QBA springs) and mixing between mineral and other waters would thus be reduced. In spring, following the winter peak in rainfall, mixing would be expected to increase. If DOM content of the shallow or surface waters exceeded that of the mineral water increased mixing would logically increase DOM content of the final discharge.

The relatively shallow QBA and OBA fresh waters contained twice the DOM concentration of mineral waters in autumn, the proposed time of minimum mixing. Surface waters were not routinely tested but in January 1998 Spring Creek contained 8 mg DOM L^{-1} and shallow riparian groundwater up to 14 mg DOM L^{-1} (Table 5.4), which were 2-3 times the average DOM content of mineral water samples. These data suggest mineral water in deep circulation is low in DOM, possibly lower than the autumn discharge value of 2 mg DOM L^{-1} as some mixing may still occur. Surface water and shallow groundwater are relatively high in DOM and may mix with ascending mineral water to increase both flow and DOM content.

Riparian zone transect

Spring water from QBA Spring 1 moved toward Spring Creek via sub-surface flow through riparian sediments. The ground water was bedrock constrained with the

bedrock profile closely resembling the water table profile but 20-25 cm deeper. Spring water at head on the floodplain surface was thus travelling laterally to the stream rather than vertically. The water underwent considerable increase in DOM content during the subsurface passage through riparian sediments (Table 5.4) indicating that DOM was leached from riparian sediments and delivered to Spring Creek.

Organic matter rich sediments immediately underlying the ponded spring water may be a source of DOM (Thurman 1985, Bishop *et al.* 1994, Dalva and Moore 1991) that would explain the two fold difference in DOM content between surface ponded spring water and ground water a short distance away. The absence of notable organic matter accumulations in the mineral dominated sediments between 9 and 5.25 m from the stream may explain why DOM content in the groundwater was the same in the three wells furthest from the stream. The wells closest to the stream were characterised by darker auger spoil with a notable presence of riparian vegetation roots. The well closest to the stream was augured through dense willow root mat. DOM leaching from live roots and decomposition of dead roots and other organic matter stored in the willow root mat could explain the increase in DOM content of water in wells closest to the stream.

DOM would also arrive at the surface of the riparian zone from canopy throughfall and stemflow (Dalva and Moore 1991, Thurman 1985). It is possible these flows infiltrate to the shallow water table thus contributing DOM. This is particularly likely where soil infiltration rates are high and topography flat, situations typical of riparian floodplains.

The relative proportions and temporal variation in the contribution of DOM from these proposed sources were not determined, but some observations can be made. The range of DOM concentrations in riparian well water in this study was within the range reported for mineral soil interstitial water and throughfall in Dalva and Moore's (1991) review of 32 publications. They were generally lower than those reported for

organic soil horizons and peatlands and one to two orders of magnitude lower than those reported for stemflow. Throughfall and stemflow would have contributed relatively little to the riparian zone immediately prior to sampling, as this occurred in mid-summer in an El-Nino drought year. A total of 24.6 mm had fallen over two raindays five days prior to sampling, but no rain had fallen in the previous 52 days. Water deficit in the riparian soils induced by the preceding two months of drought suggests little of the isolated rainfall event would infiltrate up to 50 cm to the groundwater table. In addition DOM content increased with proximity to the stream which was qualitatively related to visible organic matter content of auger spoil. Organic matter within the riparian sediment appeared to be the source of the majority of DOM delivered to the stream by QBA spring water. This is consistent with Trumbore *et al.* (1992) who showed that differences in the DOM concentration and carbon isotopic content of groundwater and stream water could be explained by the interaction between groundwater and organic rich soil layers in the riparian zone.

Groundwater DOM contribution

The product of groundwater discharge and mean groundwater DOM content gave an estimated annual groundwater DOM contribution of 91 g AFDW m^{-2} y^{-1} (Table 5.5) to the stream corridor ecosystem. The contribution to the surface stream component of the stream corridor ecosystem was 131 g AFDW m^{-2} y^{-1}. The increase was as a result of DOM sourced from riparian sediments. The later estimate was similar to the median (133 g AFDW m^{-2} y^{-1}) groundwater DOM contribution to 14 streams in other countries reported in Webster and Meyer (1997a).

Fisher *et al.* (1998) proposed a distinction between the stream corridor ecosystem and the surface water component of that ecosystem in the context of a model describing material processing and disturbance in streams. The distinction has not been specifically addressed in stream organic matter budget studies. The present study shows that this can introduce an error of approximately 50% in the estimate of groundwater DOM contribution to the surface stream component. All other parameters in published organic matter budgets, such as litterfall, GPP, respiration,

standing crop and transport, have been measured within the surface stream component. Groundwater DOM contribution, when measured as the product of bore DOM content and base flow (e.g. Wallis 1979, Marxsen *et al.* 1997, Meyer *et al.* 1997, Newbold *et al.* 1997) is for the whole of the stream corridor ecosystem. Such estimates will not measure changes in DOM that occur as groundwater travels a short distance through riparian or hyporheic sediments to the surface stream. In the present study the riparian sediments were a source of DOM, increasing estimated groundwater DOM contribution to the stream by 44%. Findlay and Sobczak (1996) showed hyporheic sediments to be a DOM sink, retaining 57% of the DOM entering as stream water flowed through a gravel bar. The last few metres of the flow path of groundwater was disproportionately important in determining DOM contribution to the stream.

Treadwell *et al.* (1997), in the only published Australian stream organic matter budget, estimated a groundwater DOM contribution of 5507 g AFDW m^{-2} y^{-1} to Keppel Creek, a fourth order central Victorian stream. This estimate was five times larger than any other reported by Webster and Meyer (1997a), suggesting either some Australian catchments have intrinsic properties that lead to large DOM fluxes through groundwater or that the method used resulted in a large estimate independent of catchment characteristics. Keppel Creek catchment geology, climate, vegetation and landuse were very similar to those of the present study, suggesting that catchment properties cannot explain the differences in estimates of groundwater DOM contribution.

In Treadwell *et al.*'s (1997) study groundwater was assumed to contribute all flow to the surface stream, DOM was determined from samples from a single bore 1 m deep and 1 m from the stream edge and stream bed area was estimated as mean bankfull width multiplied by main channel length. Each of these methods differ from those of the present study and do so in a way that effectively increases their estimate. If in the present study, Treadwell *et al.*'s methods were used, then estimates of groundwater discharge would have been three times larger (1.5×10^4 compared to 0.5×10^4 ML

y^{-1}), the mean DOM content of groundwater would have been over three times larger (14 compared to 4 mg L^{-1}, Table 5.4) and stream bed area would have been one third the size (7.1×10^4 compared to 2.3×10^5 m^2). The resulting groundwater DOM contribution of approximately 3000 g AFDW m^{-2} y^{-1} is of comparable magnitude to Treadwell *et al.*'s estimate for Keppel Creek. This suggests that the major distinction between the DOM contribution estimated in the two studies was the method used and not ecological or catchment characteristics.

This chapter has discussed the second organic matter input to be considered in this thesis and data will be used in the preparation of partial organic matter budgets in Chapter 8. The next chapter discusses community metabolism at sites within the study catchment. Two parameters relevant to the partial organic matter budgets in Chapter 8 are included; gross primary production, an organic matter input and community respiration, an organic matter output. Community metabolism has also been used to evaluate stream 'health' (Davies 1997, Davies and Bunn 1999) and this will be discussed in relation to willow and native riparian vegetation.

COMMUNITY METABOLISM

Introduction

Organic matter generated within a community is termed autochthonous organic matter. In aquatic systems, algae and macrophytes are primary producers capable of generating autochthonous organic matter. The stream community also consumes materials such as organic matter. The rate of consumption can be indicated by respiratory rates. The resulting pattern of photosynthetic and respiratory activity is termed community metabolism. This chapter considers autochthonous organic matter inputs to reaches within the study catchment, complementing the previous two chapters that considered allochthonous organic matter inputs.

Oxygen is produced during photosynthesis and this can increase dissolved oxygen levels within water bodies. During respiration oxygen is consumed and dissolved oxygen levels within water bodies can be lowered. This is the basis of many attempts to study community metabolism in aquatic communities, pioneered by Odum (1956). In general, methods can be grouped into chamber or open water methods and are discussed in detail in Owens (1969), Bott *et al.* (1978) and Bott (1996). Open water methods measure total system metabolism but are dependent upon accurate measurement of reaeration coefficients (Bott 1996). Davies (1997) suggested that open water techniques were best suited to cleared, nutrient enriched systems, as the metabolic signal was too low and the difficulty of measuring reaeration coefficients too great in forested catchments in Australia. However, the logistics of chamber methods are complex both in construction and deployment in heterogeneous stream ecosystems (Bott *et al.* 1978, Allan 1995). As preliminary measurements within the study catchment showed a readily discernible diel pattern in dissolved oxygen using an open stream method, a method to directly measure reaeration coefficients was developed and results compared to existing reaeration models (Owens 1969). In addition, the aims of this project were best addressed by measurement of total system metabolism and hence an open water method was preferred. Australian community metabolism research has been evenly divided between open water and chamber methods (Table 6.2).

The principal controls on primary productivity differ between systems (Bott 1983). Geology, geomorphology, climate, light and nutrients influence metabolic parameters within streams (Bott 1983). Geology and geomorphology are not causally related to periphyton distribution, abundance or photosynthetic efficiency but interact with climate to influence discharge, substrate, nutrients and light. Allan (1995) lists light, nutrients, current, substrate, temperature and grazing as factors controlling periphyton distribution and abundance. Wetzel and Ward (1992) and Carr *et al.* (1997) suggest light availability is an important, if not primary, control on the rate of primary production in aquatic systems.

A positive correlation between temperature and community metabolism and primary productivity has been shown in stream ecosystem studies and models (reviewed in Bott 1983 and Carr *et al.* 1997). Stream discharge influences metabolic parameters through improved solute exchange with increasing velocity up to a threshold at which current grinding action and bed movement may reduce primary productivity (Wetzel and Ward 1992, Uehlinger and Naegel 1998). Recovery following spates may be an important control on seasonal and annual metabolic rates within stream ecosystems (Uehlinger and Naegel 1998, Cushing 1997).

Comment on experimental design, analysis and interpretation

Diel oxygen curves (upstream-downstream method modified to the single station method according to Bott 1996) were used as a measure of community metabolism. Wombat Creek native (WoCrN) and Spring Creek willow (SpCrW) sites were chosen as they shared similar catchment size (Figure 3.3), catchment geology and were both downstream of water supply reservoirs. Both these sites were on spring fed creeks and remained flowing even during the drought conditions experienced in 1997-98. All other sites with the exception of WoCrW were dry during late summer-early autumn. Approximately half of SpCrW's catchment is urbanised. Approximately half of WoCrN's catchment is agricultural. However, half of both catchments are native forest in State Forest and Reserves.

The aim of the experiments in this chapter was to quantify community metabolism under a native evergreen and exotic deciduous riparian canopy over a range of

conditions throughout a single year. The limited data were not used to statistically correlate metabolic parameters with environmental conditions. Nor were differences attributed solely to differences in riparian vegetation. Community metabolism measurements in willow and native lined reaches were not replicated and attributing differences to riparian vegetation type would not be legitimate (pseudoreplication, Hurlbert 1984). However, the sites were considered well paired in terms of water quality, hydrology, catchment size and potential landuse impacts. In addition they were considered typical of the tree-lined, mid-order reaches of the study catchment. As a result, they were studied to give indicative data for tree-lined reaches used in the construction of organic matter budgets in Chapter 8. Despite the inherent problem of lack of replication, the discussion section of this chapter (Section 6.4) advances arguments that may explain some of the differences observed between sites and also provides suggestions for further experimentation. As Underwood (1997) advises, these arguments are based on ancillary evidence, including that from the literature, and inductive reasoning based on knowledge of the canopy phenology and response of in-stream autotrophs to light.

Methods

Diel dissolved oxygen measurements and analysis

The upstream-downstream diel dissolved oxygen method (Odum 1956, Bott 1996) was used to gather data for a 70-80 m reach at the WoCrN and SpCrW sites. Consideration was given to both deciduous canopy conditions and season in determining the experimental timing, with the result that experiments were conducted at the following times: full leaf in early summer, 50% leaf in autumn, following leaf fall in early winter, and just prior to leaf-out in late winter (Table 6.1). Equipment malfunction interfered with data collection at WoCrN in late winter/spring.

Site	Date	Duration (h)
WoCrN	Dec 1997	43
	Mar 1998	42
	Jun 1998	25
SpCrW	Aug 1997	42
	Nov 1997	27
	Mar 1998	66
	May 1998	46

Table 6.1 Date and duration of community metabolism measurements at WoCrN and SpCrW sites.

Two dissolved oxygen meters (TPS model LC82 with ED500 DO_2 probe) were connected to a datalogger (Campbell Scientific 21X) recording at 10 minute intervals. Probes were positioned 70-80 m apart and placed perpendicular to stream flow at a channel constriction to ensure water velocity past the probe membrane was adequate. This was tested by rapidly stirring the probes to further increase velocity past the membrane and confirming that the meter reading was unchanged. Probes were submerged and shaded to stabilise their temperature. Continuous data were recorded for a minimum of 25 hours and up to 66 hours for any particular site-season combination (Table 6.1). Occasionally probes were disturbed or wires chewed by rodents resulting in lost data for one probe for a number of hours. As mean values were used for the single station method, the remaining probe's data were used to obtain a continuous data set.

Throughout community metabolism measurements probe calibration was checked in air according to the manufacturers instructions. The procedure, including temperature stabilisation, took 15-20 minutes thus interfering with few stream dissolved oxygen readings. DO meters were not adjusted but the magnitude of any drift was noted. The drift was recorded as the percent saturation read by the meter in air (probe should read 100% saturation in air). If the DO meter read 95% in air, a value recorded by the datalogger at that time would need to be increased by 1.053 (ie. 1/0.95) to reflect the true level of dissolved oxygen in the stream. Similarly if the DO meter read 103% the corresponding datalogger value would need to be reduced by a factor of 0.971 (ie. 1/1.03). Down-loaded data were corrected for calibration drift by assuming any drift

observed occurred uniformly over the time period between calibration checks. A sub-routine within a spreadsheet allowed the data to be transformed rapidly, such that:

$$DO_{(corrected,\ T\alpha)} = DO_{(raw,\ T\alpha)} \times (T_i + (D_{i+1} - D_i / T_{i+1} - T_i) \times (T\alpha - T_i)$$

Where:

$DO_{(corrected,\ T\alpha)}$ = dissolved oxygen at time α ($T\alpha$) corrected for calibration drift, such that $T\alpha$ is between the i^{th} and $i^{th}+1$ calibration checks

$DO_{(raw,\ T\alpha)}$ = dissolved oxygen at $T\alpha$ prior to correction for calibration drift

T_i = time of the i^{th} calibration check, in minutes since beginning of experiment

T_{i+1} = time of the $i^{th}+1$ calibration check, in minutes since beginning of experiment

D_i = drift at time of i^{th} calibration check, as inverse of percent saturation recorded by DO meter with probe in air

D_{i+1} = drift at time of $i^{th}+1$ calibration check, as inverse of percent saturation recorded by DO meter with probe in air

$T\alpha$ = time, in minutes since beginning of experiment, of the α dissolved oxygen reading

The first reading of the experiment ($T_\alpha = 0$) requires no calibration correction as probes were calibrated immediately prior to each experiment, thus the above routine was not used as it would yield a value of zero. Similarly, no data for the stream were being collected during a calibration check so the expression $T\alpha - T_i$ did not need to be evoked as this would again yield a zero result.

The corrected data showed minor differences between upstream and downstream sites for some experiments. These were analysed by the upstream-downstream method and results presented in Wilson (1999). Smoothing of data from 10 minute samples to hourly means as recommended by Bott (1996), was undertaken to independently reduce variability in upstream and downstream data. These data were then analysed by the upstream-downstream method. However, when the smoothing technique was combined with thorough analysis of temperature and re-aeration coefficients (Section 6.2.2) upstream-downstream differences were considered minor or absent. Thus diel

dissolved oxygen patterns across the experimental reaches appeared uniform. In addition, riparian vegetation and surrounding catchment conditions were uniform for kilometres upstream and downstream of the experimental reaches. These characteristics of both the data and the streams meant the single station method was appropriate (Odum 1956, Bott 1996). Data were re-analysed according to the single station method (Bott 1996) using the mean of the upstream and downstream dissolved oxygen readings for each time interval. Hourly means were generated from the 10 minute interval data to facilitate the precise use of Bott's (1996) spreadsheet and formulas.

Mean depth of the 70-80 m experimental reach was used to obtain metabolic parameters in areal terms ($gO_2\ m^{-2}\ d^{-1}$, Bott 1996).

Reaeration coefficient

Where stream depth and velocity were within appropriate ranges, reaeration coefficients ($f_{20°C}$, $m\ h^{-1}$) were determined from equations derived by Owens (1969, Owens *et al.* 1964) and summarised in Bott (1996). These estimates are based on mean stream velocity and depth (velocity-depth method).

A direct measure of reaeration was also made, based on that of Grimm and Fisher (1984). A sheet of plastic, initially 10 m x 2m then increased to 20 m x 4 m, was laid over the flowing stream with the edges of the plastic rising a short distance up the channel bank and weighted with stones (Figure 6.1). The upstream edge of the plastic was then buried in the stream sediment diverting the stream flow over the plastic.As the plastic was molded to the stream channel by the weight of the stream water, morphology of the channel was not affected by this manipulation. Benthic and periphyton communities under the plastic no longer contributed to respiratory and photosynthetic changes in dissolved oxygen within the water column. These communities, rather than phytoplankton, dominate autotrophic and heterotrophic activity in low order forested streams (Chessman 1985, Bott 1996) thus changes in dissolved oxygen observed between the upstream and downstream ends of the plastic sheet were assumed to be a result of reaeration alone.

Figure 6.1. Plastic lined reach at WoCrN used for reaeration coefficient measurements. Flow was isolated from the benthic community and thus measured change in dissolved oxygen between upstream and downstream ends of the plastic sheet was attributed to diffusion. Plastic sheet was 4 m wide and surface stream approximately 2 m wide as an indicative scale.

Flow was allowed to stabilise for a time that exceeded the retention time of the 10 m or 20 m plastic lined reach. Reaeration rate was measured as the difference between the dissolved oxygen content of the stream water at the upstream and downstream edges of the plastic (converted to g O_2 m^{-2} h^{-1}).

Temperature was recorded by thermometer at the upstream end, middle and downstream end of the plastic and the mean temperature used in calculations. The reaeration coefficient at the experimental temperature ($f_t°_C$) was calculated by dividing the reaeration rate (gO_2 m^{-2} h^{-1}) by the mean saturation deficit (g m^{-3}) across

the plastic lined reach (Grimm and Fisher 1984). The reaeration coefficient at 20°C was determined using:

$$f_{20°C} = f_{t°C} / 1.024^{(t-20)} \qquad \text{(Bott 1996).}$$

As the length of plastic was approximately 25% of the length of the reach used for dissolved oxygen measurements a conversion factor was determined using the velocity-depth method. A reaeration coefficient for the 70-80 m stream reach used for dissolved oxygen measurements was determined. The ratio of velocity-depth reaeration coefficients for the two reaches (the 10-20 m direct method reach and the 70-80 m dissolved oxygen reach) was used as a conversion factor for the directly measured reaeration coefficient. The converted directly measured reaeration coefficient was used for all subsequent calculations as it could be determined under low flow conditions that were outside the range applicable to the velocity-depth method.

Photosynthetically active radiation

Experiments were conducted during stable weather patterns with generally cloudless skies. As the measurements at the two sites were not conducted simultaneously, this precaution avoided dramatic differences in light levels incident on the riparian forest canopy. Consequently, comparative light levels at the creek surface between sites and within any particular season related to canopy characteristics rather than variation in incident light. Incident light on the canopy was seasonally determined with higher light intensity on clear days in summer than in winter.

Light levels (photosynthetically active radiation, PAR, in $\mu E\ m^{-2}\ s^{-1}$) incident on the water surface were recorded with a Sunfleck Ceptometer (Decagon Devices, model SF80). To address the variability of light levels under a canopy the SF80 has 80 light sensors placed at one centimetre intervals and the average of these sensors was recorded for each reading. Readings were taken immediately above the mid-stream water surface at 5m intervals along the 70-80 m experimental reach (approximately 15-17 readings). Readings were taken approximately hourly throughout the day for each experiment and the times recorded as Eastern Standard Time (EST) not as daylight saving time. A reference reading was taken immediately prior to creek

sampling from unshaded and cleared adjacent ridges. This was assumed to be very similar to the light levels incident on the canopy of the riparian forest. The experimental reach values were expressed as the percentage of PAR at the unshaded site. The mean percentage for daylight hours was used as an indicator of canopy density ('canopy gap fraction', Martens *et al.* 1993), where 100% would indicate no canopy and 0% a completely opaque canopy. This is equivalent to Pidgeon's (1978) shading factor. However, it is preferable to consider the inverse of canopy gap fraction as a canopy shading factor.

Canopy gap fraction can be used to derive estimates of leaf area index (LAI) but Martens *et al.* (1993) showed that without subtraction of diffuse radiation LAI was overestimated. In addition a choice of models for analysing the data must be made and the choice influences the magnitude of derived LAI (Martens *et al.* 1993). Thus it was deemed preferable to present canopy gap fraction data, thus allowing comparison of sites without the complication of model choice and associated assumptions.

Temperature, discharge and stream dimensions

Temperature was recorded throughout the day with hand thermometers and pre-dawn minimum and pre-dusk maximum temperatures recorded with a maximum-minimum thermometer. All data for each experiment were analysed with a curve fitting program (KURV, Conrad Buttons Software). The resulting polynomial gave the expected temperature as a function of time since the beginning of the experiment and this formula was inserted in the temperature column of the analysis spreadsheet to give the expected temperature for each 10 minute interval.

Discharge was measured by capturing the total stream flow in a 20 L plastic bag over a timed period between two and ten seconds. At SpCrW high stream flows were captured at a small (<0.5 m) waterfall a short distance upstream of the experimental reach. Low flows were captured at an artificial 'waterfall' created by a 400 mm wide piece of metal forced into the substrate across the stream channel where it flowed down a sloping section of willow root mat. This created a 'lip' over which the water flowed and could be collected. As the metal was forced approximately 100 mm into the porous root mat some of the subsurface flow was captured. At WoCrN, flows

were captured during reaeration coefficient measurements. All stream flow was diverted over the plastic sheet (Section 6.2.2) and at a slight constriction in the stream channel a lip was formed by bunching the plastic into a small step. The plastic bag was then used to capture stream flow over a timed period.

Stream dimensions were recorded at approximately 1 m intervals between the upstream and downstream dissolved oxygen probes. Stream width (W) and depth (D) at the deepest point were recorded for each interval. Stream cross sectional shape was triangular at most locations and this was the assumption used to calculate stream cross sectional area ($0.5W \times D$) for each interval. A section of the experimental reach was bounded by successive intervals and the volume of each section was calculated by multiplying stream cross sectional area by section length. The latter was estimated from total reach length divided by the number of measurement intervals. Total volume of the experimental reach was the sum of section volumes. Surface area was calculated for each section as width times section length and summed for the total reach. Flow ($m^3 s^{-1}$) and volume (m^3) were used to calculate hydraulic retention time.

Data analysis

Means of GPP, CR_{24}, NDM and P/R were compared between sites using two-tailed t-tests for independent samples (Statistica, StatSoft Inc). Canopy gap fraction was compared between season and site, with their interaction, using ANOVA (Statistica, StatSoft Inc). A conservative *post hoc* test was used (Duncan's multiple range test). Post hoc power analysis was conducted using GPOWER (Faul and Erdfelder 1992). As 'treatments' (willow and native-lined reaches) were not replicated significant differences between sites were not solely attributed to canopy conditions.

Results

Dimensions, temperature and discharge

WoCrN and SpCrW showed similar seasonal patterns in dimension, temperature and discharge (Table 6.2). Dimensions, temperature and discharge were greatest in early summer. Temperatures were lowest in winter. Dimensions and discharge were lowest in autumn. For each season, dimensions and discharge were lower at WoCrN despite

similar catchment size (Table 3.1). However, these parameters were not continuously monitored and dates of experimentation at the two sites were not identical.

	WoCrN				SpCrW				
	Dec 1997	Mar 1998	Jun 1998	Mean	Aug 1997	Nov 1997	Mar 1998	May 1998	Mean
T (°C)	14.8	10.8	4.9	10.2	4.6	16.7	12.6	7.2	10.3
D (m)	0.23	0.16	0.2	0.2	0.26	0.29	0.20	0.27	0.26
W (m)	2.03	1.54	1.89	1.82	2.19	2.15	1.08	2.10	1.88
Q (Ls^{-1})	4.7	0.3	1.2	2.07[a]	11.8	22.5	0.20	7.6	10.5[a]
GPP ($gO_2\ m^{-2}\ d^{-1}$)	1.08	1.19	0.16	0.81	0.48	0.56	0.63	0.44	0.53
CR_{24} ($gO_2\ m^{-2}\ d^{-1}$)	2.16	8.40	0.78	3.78	0.96	4.56	2.64	2.16	2.58
NDM ($gO_2\ m^{-2}\ d^{-1}$)	-1.09	-7.21	-0.62	-3.00	-0.49	-4.00	-2.01	-1.72	-2.06
P/R	0.50	0.14	0.21	0.21	0.5	0.12	0.24	0.20	0.21

Table 6.2 Environmental and metabolic parameters determined for WoCrN and SpCrW sites between August 1997 and June 1998. T, mean daily temperature; D, mean depth; W, mean width; Q, discharge; GPP, gross primary production; CR_{24}, 24 hour community respiration; NDM, net daily metabolism; P/R, ratio of GPP to CR_{24}. [a] these values can be compared to higher estimates of annual discharge in Table 3.1 from Nathan and Weinmann (1993). Figures in this table were not derived from continuous monitoring and spates were avoided during community metabolism experiments. As stream dimensions relate to discharge these comments also apply to data for D and W.

Photosynthetically active radiation

Midday photosynthetically active radiation recorded at unshaded sites adjacent to experimental reaches showed an expected seasonal pattern with summer levels 2.4 times winter levels (2000 and 830 $\mu E\ m^{-2}\ s^{-1}$ respectively). The canopy gap fraction (daily mean of hourly measurements of PAR at the creek surface as a percentage of PAR at an adjacent unshaded site) showed contrasting seasonal patterns at the two sites (Figure 6.2). The canopy gap fraction at WoCrN was greatest in summer and lowest in winter, with intermediate values in spring and autumn ($F_{3,353} = 10.1$, $P<0.001$). The canopy gap fraction at SpCrW was greatest in winter and least in summer ($F_{3,693} = 43.6$, $P<0.001$). There were significant differences between the sites in all seasons (all $P<0.001$, Duncan's *post hoc* test). However there was little difference in the range of canopy gap fractions recorded at each site (6-20% and 7-26% at WoCrN and SpCrW respectively, Figure 6.2).

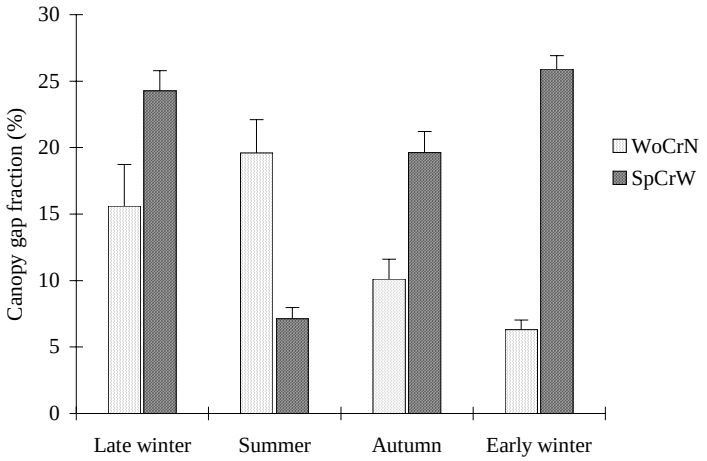


Figure 6.2. Canopy gap fraction (mean of daytime hourly measurements, error bars denote one standard error) for WoCrN, a reach lined by native, evergreen riparian forest, and SpCrW, a reach with deciduous willow riparian forest.

Seasonal variation in canopy penetration and incident (unshaded) radiation combined to give a more stable light environment under the deciduous canopy at SpCrW with midday radiation at the creek surface ranging from 145-375 μE m^{-2} s^{-1} compared to 38-568 μE m^{-2} s^{-1} under the evergreen canopy at WoCrN. The greatest midday light intensity at the creek surface was recorded in summer at WoCrN and autumn at SpCrW. Lowest midday light intensity at the creek surface was recorded in winter at WoCrN and summer at SpCrW.

Gross primary production (GPP)

GPP ranged from 0.16 to 1.19 g O_2 m^{-2} d^{-1} at the two sites (Table 6.2, Figure 6.3). A seasonal pattern was not marked at SpCrW, with only slightly greater GPP in summer and autumn relative to early and late winter (Figure 6.3). GPP at WoCrN in winter was much lower than in summer and autumn giving a distinct seasonal trend. GPP at SpCrW appeared to be approximately 50% of that recorded at WoCrN in summer and autumn but 2.75 times larger in late autumn – winter. Estimated mean GPP was 0.81 and 0.53 g O_2 m^{-2} d^{-1} at WoCrN and SpCrW respectively (Table 6.2). Estimated means for the two sites were not significantly different ($t = 1.02$, 5 degrees of freedom, $P = 0.36$), but effect size (0.73) was medium and power (0.12) low (Cohen 1988).

Twenty four hour community respiration (CR_{24})

CR_{24} ranged from 0.7 to 8.4 g O_2 m^{-2} d^{-1} at the two sites (Table 6.2, Figure 6.3). Rates were least in winter at both sites but greatest in autumn at WoCrN and summer at SpCrW. CR_{24} was not consistently higher at one site relative to the other but the range of values was greater at WoCrN giving a stronger seasonal trend. Estimated mean CR_{24} was 3.78 and 2.58 g O_2 m^{-2} d^{-1} for WoCrN and SpCrW respectively (Table 6.2). Estimated means for the two sites were not significantly different (t = 0.56, 5 degrees of freedom, P = 0.6) and both effect size (0.24) and power (0.1) were low (Cohen 1988).

Net daily metabolism (NDM) and ratio of GPP to CR_{24} (P/R)

GPP and CR_{24} subtract to give NDM and divide to give P/R. NDM was always negative and ranged from –0.49 to –7.21 g O_2 m^{-2} d^{-1} at the two sites. Estimated means were –3.0 and –2.06 g O_2 m^{-2} d^{-1} at WoCrN and SpCrW respectively and were not significantly different (t = 0.47, 5 degrees of freedom, P = 0.66). P/R ratios were always ≤0.5 ranging from 0.12 to 0.5 at the two sites. Estimated means for the two sites were not significantly different (t = 0.14, 5 degrees of freedom, P = 0.9) and both effect size (0.16) and power (0.1) were low (Cohen 1988).

As the two parameters, NDM and P/R, are derived algebraically from GPP and CR_{24} seasonal patterns are similar with low (more negative) values of NDM corresponding to low P/R ratios. The seasonal trends of NDM and P/R were similar and generally mirrored that of CR_{24}. At WoCrN, NDM and P/R were lowest in autumn and greatest in winter and summer respectively. At SpCrW, NDM and P/R were lowest in summer and greatest in late winter

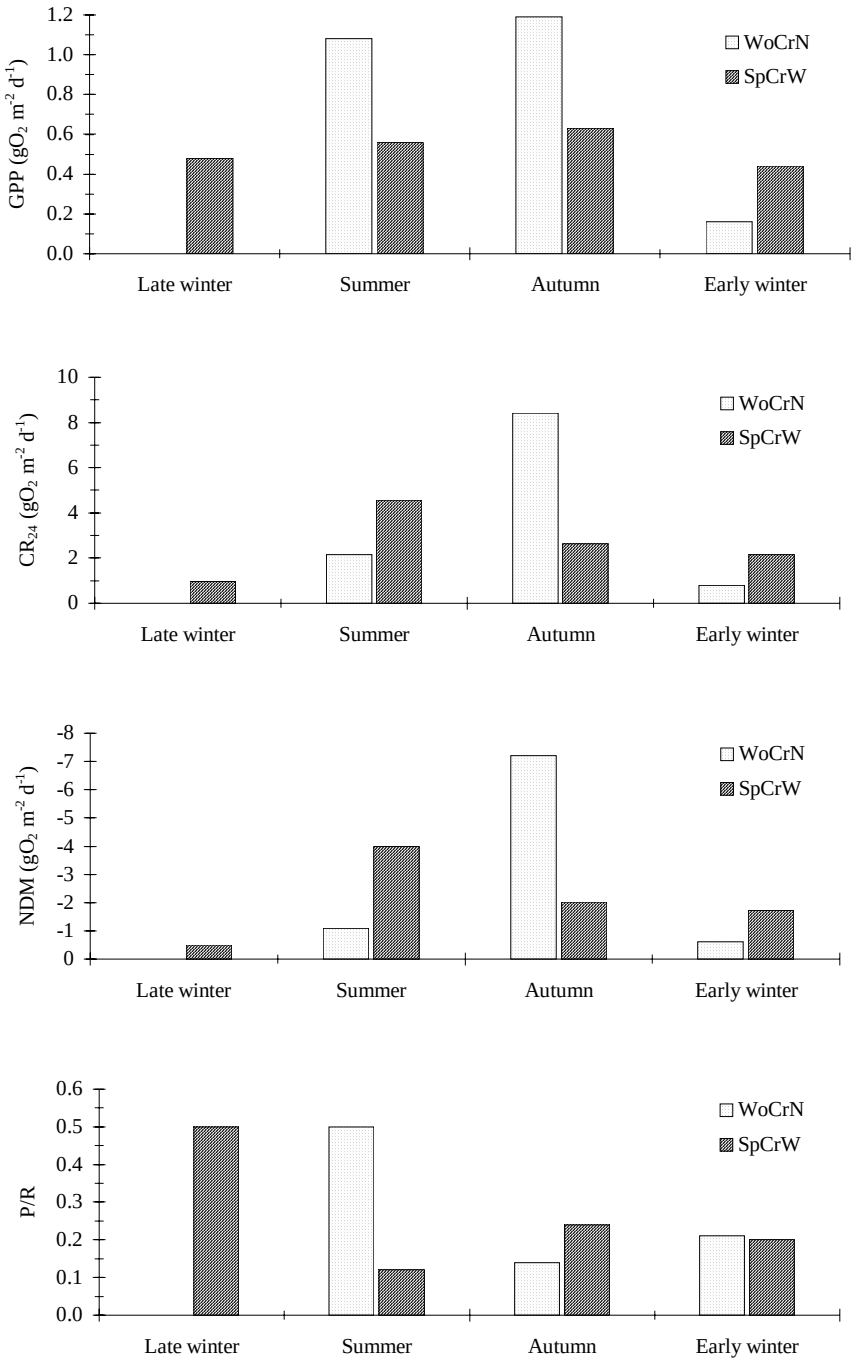


Figure 6.3. Gross primary production (GPP), 24 hour community respiration (CR_{24}), net daily metabolism (NDM) and GPP to CR_{24} ratio (P/R) for WoCrN and SpCrW. Estimates from single station open water diel dissolved oxygen curves.

Discussion

Environmental conditions

The two sites, WoCrN and SpCrW, were very similar in mean temperature, depth, width and annual range of canopy gap fraction (Table 6.2). However, discharge appeared to be many times greater at SpCrW and estimates from both sites were much lower than the mean discharge estimates in Table 3.1 predicted from Nathan and Weinmann (1993). The reason for the apparent discrepancy in discharge between Table 6.2 and Table 3.1 is that the data collected during community metabolism experiments were not continuous and spates were avoided during community metabolism experiments. This can be compared to the model of Nathan and Weinmann (1993) which was based on continuous monitoring of discharge over three and a half decades. In addition 1997 and 1998, the years of the present study, were drought years with below average rainfall.

The draw-down characteristics of the water supply reservoirs upstream of the two sites may explain the lower than expected discharges at WoCrN. Wombat Reservoir, upstream of WoCrN, was used to supply reticulated water to Daylesford and Hepburn during the period of study, whereas no extractions were made from Hepburn Reservoir upstream of SpCrW. Central Victoria has a winter dominated rainfall pattern and in summer constant extraction from Wombat Reservoir during periods of little rain depleted the Reservoir leaving it at approximately 50% capacity in autumn. The influence of the 'autumn break' (the beginning of the winter period rainfall) was observable in the marked increase in dimension and discharge data for SpCrW between March 1998 and May 1998 (Table 6.2). The effect was much less significant at WoCrN suggesting that reservoir refilling impacted on downstream discharge.

Seasonal changes in stream temperature generally follow seasonal trends in mean monthly air temperatures (Allan 1995). This appears to effectively explain the seasonal variation in temperature observed in the present study (Figure 3.2 and Table 6.2). Temperature data were not collected simultaneously from the two sites. However, community metabolism experiments at the two sites were within two weeks of each other for each season. It appears that SpCrW was 1-2 °C warmer than WoCrN on each occasion, possibly a result of rock outcrops absorbing solar energy

upstream of the study reach. The stream flowed across the outcrops, which had bedding planes approximately perpendicular to the stream flow. Near WoCrN the creek was flowing parallel to the main bedding planes of the Ordovician strata and bedrock outcrops crossing the stream path were absent. During sunny days shallow slow moving pools perched on these outcrops contained measurably warmer water than pools upstream and downstream with silty substrate. The bedrock outcrops appeared to also influence the cover of riparian vegetation, presumably limiting the rooting volume available for trees in the vicinity of the outcrops. The combination of greater light penetration of the sparse canopy and heating of the dark coloured rocks may have contributed to relatively warm water in Spring Creek.

Light levels incident on the riparian canopy showed a predictable seasonal pattern, with maximum intensity in summer and minimum in winter (midday intensities approximately 2000 and 830 $\mu E\ m^{-2}\ s^{-1}$ respectively). For the deciduous canopy at SpCrW the seasonal pattern in canopy gap fraction was matched to leaf fall and leaf out, being least in summer and greatest in winter. The interaction between incident light and canopy gap fraction resulted in maximum midday light levels at the creek surface in autumn and minimum levels in summer. It is probable the midday light intensity would have been even higher immediately following full leaf fall, which occurred soon after the autumn measurements.

A contrasting seasonal pattern in canopy gap fraction was observed at WoCrN, with a maximum in summer and minimum in winter. This pattern complemented the seasonal pattern of incident light intensity resulting in high midday light intensities at the creek surface in summer and very low intensities in winter (568 and 38 $\mu E\ m^{-2}\ s^{-1}$ respectively). The seasonal pattern of canopy gap fraction for the deciduous canopy was easily explained by leaf fall in autumn and leaf out in spring. However, the seasonal pattern observed at WoCrN was of similar magnitude but was not accompanied by obvious canopy thinning.

If the canopy composition and structure did not change as in the deciduous canopy (ie. no complete leaf fall) then the difference in the proportion of light intercepted may be related to characteristics of the incident light. Campbell (1986) showed that

the extinction coefficient for a canopy could be expressed in terms of the angle of incident light (zenith angle being the angle measured from the vertical) and a parameter x. The parameter x could be measured as the ratio of the projected area of an average canopy element on a horizontal plane to its projection on a vertical plane. Where leaves hang vertically x approaches zero and the extinction coefficient of a canopy becomes larger with decreasing zenith angle of incident light. Eucalypt leaves typically hang near vertically and thus the canopy extinction coefficient (ie. the probability of light penetrating a canopy without interception) is smallest in summer when the sun is highest in the sky (smallest zenith angle) and largest in winter when the sun is low in the sky. Midday canopy gap fractions observed at WoCrN in summer and winter could be explained by changes in sun angle alone if the parameter $x \approx 0.75$ and LAI remained constant. A value of $x \approx 0.75$ would mean that, on average, each canopy element is hanging a little below 45°, which appears appropriate for a eucalypt dominated forest.

Gross primary production

Lack of spring data for WoCrN and lack of replication of sites and seasons behoves a conservative approach to the community metabolism data. However, at the catchment level the data represent nearly 300 hours of monitoring over an 11 month period at two 70-80m experimental reaches. Thus they indicate the magnitude of metabolic parameters within streams lined by the dominant riparian vegetation types within the catchment.

Estimates of mean GPP were comparable with data from similar sized Australian streams (Table 6.3). Davies (1993, unpublished thesis cited in Treadwell 1995) and Treadwell (1995) recorded low rates of GPP (0.1 and 0.04 g O_2 m^{-2} d^{-1}, respectively) in heavily shaded, low nutrient, low order headwater streams in Western Australia and Victoria. GPP similar to rates measured in the present study were determined by Chessman (1985) at five locations along the La Trobe River in Victoria and by Davies and Bunn (1999) for forested streams in Queensland and Western Australia. Larger rates of GPP have been recorded for fourth to sixth order streams of the Acheron Basin in central Victoria (Treadwell 1995), third order reaches of Commissioners Waters in the New South Wales Tablelands (Pidgeon 1978) and in

the Pilbara region of Western Australia (Davies 1997). Rates of GPP up to 18 g O_2 m^{-2} d^{-1} (20-30 times those of the present study) have been recorded on the fifth order Little River, Victoria (Treadwell 1995) and in the Pilbara, WA (Davies 1997). These are the highest rates recorded in Australia.

Light, nutrients and stream size are factors that may broadly explain differences observed throughout Australia in GPP. In a study of 30 streams from the USA and other countries, Lamberti and Steinmann (1997) used linear and multiple regression analysis to resolve relationships between GPP (the dependent) and 13 predictor (independent) variables. Three predictors, watershed area, discharge and soluble reactive phosphorus (SRP), could explain seventy percent of the observed variation in GPP. In the linear regressions, percent canopy cover was inversely related to GPP and explained 29% of the variation.

Australian studies of stream GPP appear consistent with both the range of values and the regression analysis in Lamberti and Steinmann (1997). Small watershed area, dense riparian vegetation cover and low nutrient status could explain very low rates of GPP recorded in Australia (Table 6.3). Increasing watershed area and canopy openness largely explained increasing rates of GPP with increasing stream order in low- to mid-order reaches of a river continuum in the studies of Treadwell (1995) and Chessman (1985). Both these studies found rates of GPP to be lower at the most downstream sites studied than at mid-reaches, an observation consistent with the River Continuum Concept (Vannote *et al.* 1980).

A few Australian studies appear to have recorded rates of GPP that are noticeably higher than those reported in the present study for apparently similar sized streams. Pidgeon (1978) studied a third order stream that included a willow lined reach. He reports a watershed area of 378 km^2 compared to 17 km^2 for the fourth order streams in the present study. Grazing animals had access to the stream and phosphate–P was equivalent to total phosphorus in the present study, suggesting nutrient enrichment. The willow lined reach had a canopy gap fraction of 20-30 % compared to 7-26 % for SpCrW in the present study. These differences could explain the two- to four-fold greater GPP recorded by Pidgeon compared to the present study.

Extremely high light inputs and livestock impacts, presumably nutrient enrichment, were considered key factors by Davies (1997) in explaining the very high rates of GPP recorded in the Pilbara region of Western Australia. None of these factors seems to explain the disparity between Treadwell's (1995) results for Keppel Creek and the present study. Keppel Creek and WoCrN had very similar riparian vegetation species composition, stream order (fourth), watershed area (14.3 and 16.5 km^2 respectively), gradient (0.027 and 0.019 m/m), bankfull dimensions and nutrient status (NO_x = 0.07-0.14, P_{total} =0.01 and NO_x = 0.2, P_{total} = 0.01 mg L^{-1}). Discharge (70-850 and 88 L s^{-1} respectively) and canopy gap fraction (10-30 and 6-20%) appeared to be slightly greater at Keppel Creek. Despite these seemingly minor differences Treadwell recorded rates of GPP 3.5 times larger than those recorded at WoCrN.

The difference is probably not explained by differences in method. Bott *et al.* (1978) in a comparative study of five methods for measuring community metabolism suggested open stream methods yielded estimates of GPP that compared favourably with one or more chamber methods. Indeed the chamber method most similar to that used by Treadwell (1995) yielded lower estimates of GPP than the open water method.

Open water methods are complicated by the accurate determination of reaeration coefficients (Bott *et al.* 1978, Bott 1996, Section 6.2.2). It is worthwhile considering the consequences for GPP estimates of errors in estimating reaeration coefficients. In the present study two methods were used to determine reaeration coefficients, a direct measure using plastic sheeting and Owen's (1969) stream velocity-depth method. The two methods yielded similar results at both sites under a range of flow conditions. The direct method was 60-85% of the velocity-depth method, with the exception of a single reaeration experiment performed on a 10 m riffle at SpCrW where the direct method was only 13% of the velocity-depth method. The similarity in estimates generally obtained by the two methods suggests the reaeration coefficients used in the present study are reliable.

Author	Location	Method	Canopy	Order	GPP ($g\ O_2\ m^{-2}\ d^{-1}$)	CR_{24} ($g\ O_2\ m^{-2}\ d^{-1}$)	NDM ($g\ O_2\ m^{-2}\ d^{-1}$)	P/R
The present study	Wombat Cr. VIC	Single station	Native forest	4	0.78 (0.16-1.19)	3.78 (0.78-8.4)	-3.00 (-0.62 to -7.21)	0.21 (0.14-0.5)
	Spring Cr., VIC		Willow	4	0.53 (0.44-0.63)	2.58 (0.96-4.56)	-2.05 (-0.49 to –4.0)	0.21 (0.12-0.5)
Pidgeon 1978	Commissioners Waters, NSW	Single station	Open eucalypt woodland	3	3.54 (2.86-4.78)	2.90 (2.14-3.78)	0.64	1.22 (1.18-1.26)
			Cleared	3	2.15 (1.39-2.61)	1.98 (1.12-2.55)	0.17	1.09 (0.99-1.25)
			Willow	3	1.13 (0.57-1.9)	1.27 (0.9-1.82)	-0.13	0.89 (0.61-1.05)
Chessman 1985	La Trobe R, VIC	Single station	Native forest	15 km[a]	0.15	2.97	-2.82	0.05
			Native forest	48 km	0.59	4.61	-4.02	0.13
			Willow	76 km	0.23	4.13	-3.90	0.06
			Sparse - agricultural	111 km	1.90	3.78	-1.88	0.50
			Sparse - agricultural	172 km	0.74	2.81	-2.08	0.26
Davies 1993, cited in Treadwell 1995	Nth Dandalup, WA	Open chamber	Native forest	1	0.11-0.69	0.19-1.04		<1
			Agricultural	4	0.002-0.71	0.006-1.02		<1
Treadwell 1995	Tweed Spur Cr., VIC	Open chamber	Native forest	2	0.04 (0.014-0.053)	0.19 (0.15-0.21)	-0.14 (-0.1 to –0.17)	0.20 (0.09-0.29)
	Keppel Cr., VIC		Native forest	4	2.79 (1.53-4.0)	5.0 (3.13-6.45)	-2.22 (-0.74 to –3.24)	0.57 (0.35-0.76)
	Little R, VIC		Native forest	5	13.07 (7.94-17.87)	19.66 (15.32-22.88)	-6.59 (-2.91 to –9.48)	0.66 (0.52-0.86)
	Acheron R, VIC		Native/willow forest	6	2.74 (0.13-5.72)	10.25 (8.93-10.98)	-8.05 (-5.13 to -10.41)	0.27 (0.01-0.53)
Davies 1997[bc]	Pilbara, WA	Open chamber	MRHI reference	-	6.4	5.0	1.4	1.3
			MRHI monitoring	-	17.9	11.0	6.9	1.6
Bunn *et al.* 1997	Bamboo Cr., QLD	Single station	cleared	4	-	-	-	0.68-1.27
Davies and Bunn 1999[bd]	Johnstone R, QLD	Open and closed chambers	Native forest	-	0.58 (0.52-0.63)	1.01 (0.83-1.19)	-0.43	0.57 (0.52-0.62)
	Mary R, QLD		Native forest	-	0.49	0.55	-0.06	0.89
	Jarrah forest, WA		Native forest	-	0.34 (0.28-0.4)	0.67 (0.53-0.8)	-0.33	0.51 (0.49-0.51)

Table 6.3. Summary of community metabolism data for Australian streams and rivers. All studies used diel dissolved oxygen changes to estimate metabolic parameters using the open water single station or chamber methods (Bott 1996). Values represent an annual mean derived from the mean of seasonal values, with the range in seasonal value in parentheses. [a] values represent distance from source along a river continuum, stream order not given. [b] data converted using photosynthetic quotient of 1.2, respiratory quotient of 0.85 and 1 mg C = 2.667 mg O_2. [c] metabolic parameters estimated from a graph as tabulated data not given. [d] summer and winter for Johnstone R and the Jarrah forest, winter only for Mary R.

In addition, the error induced in estimates of community metabolism as a result of an error in reaeration coefficient can be determined. The magnitude of the rate of change in dissolved oxygen due to reaeration can be compared to that due to community metabolism (Bott 1996). In the present study dissolved oxygen in the water column was always less than 100% thus oxygen always diffused from air to water. The mean daily increase in stream dissolved oxygen due to reaeration was between 45 and 100% (mean 82%) of the mean daily decrease due to community respiration. As the magnitude of the two signals is similar an error in reaeration coefficient would yield a similar magnitude error in community metabolism. The impact on GPP of a change in reaeration coefficient was generally less since the plot used to estimate GPP and CR_{24} remained much the same shape but moved up and down with respect to the vertical axis. In the present study a doubling of reaeration coefficient would lead to changes in GPP and CR_{24} as summarised in Table 6.3

		CR_{24}	GPP
WoCrN	Dec 1997	x 1.9	x 1.5
	Mar 1998	x 2.0	x 2.0
	June 1998	x 1.9	x 1.2
SpCrW	Aug 1997	x 1.5	x 1.2
	Nov 1997	x 1.9	x 1.04
	Mar 1998	x 2.1	x 1.5
	May 1998	x 2.0	x 1.3

Table 6.4. Increase in CR_{24} and GPP (figure represents a multiplication factor) induced by arbitrarily doubling the reaeration coefficient for all community metabolism experiments performed in the present study.

It appears that errors in reaeration coefficient would need to be many fold to explain the difference in GPP estimates at Keppel Creek and WoCrN. This would seem unlikely given the similarity in the direct and velocity-depth methods used to determine reaeration coefficients. The reasons for the difference in GPP between Keppel Creek and WoCrN deserve elucidation, as their similarity in form and geographic proximity suggests such a study could improve understanding of methodology, biology and energetics in an Australian context.

Mean GPP was not significantly different between the two sites in the present study, but statistical power was low due to both lack of replication and low sample size. However, the effect size was moderate (0.73) suggesting that experimental designs with increased power could be revealing (Aron and Aron 1997). For community metabolism experiments power could be effectively increased through increased sample size and more precise measurement available through recent technological advances in dissolved oxygen probes which would effectively reduce the sample standard deviation.

WoCrN appeared to have greater rates than SpCrW in summer and autumn but lower in winter. Lamberti and Steinman (1997) showed percent canopy cover to be significantly and negatively correlated with GPP, explaining 29% of the variation in GPP in 30 lotic systems across the globe. From this, a speculative association between the light climate and GPP under the two canopies could be presented as an explanation for the observed patterns in the present study. The bright and warm conditions of summer and autumn could be favourable to photosynthesis under both canopies. The brighter conditions at WoCrN over summer may allow a relatively strong autotrophic community to develop and continue to be productive in autumn despite decreasing light and temperature. In winter at WoCrN the autotrophic community may well be light limited with very low levels of PAR even at midday (38 $\mu E\ m^{-2}\ s^{-1}$). The more stable seasonal pattern of PAR at SpCrW may allow a less marked seasonal pattern in GPP. At SpCrW slightly higher temperatures and discharge may compensate for the relatively low light conditions in summer. Limiting factors for primary productivity in streams include light, discharge, temperature, grazing and availability of nutrients (Bott 1983, Allan 1995, Lamberti and Steinmann 1997, Giller and Malmqvist 1998) and, given their inevitable interaction over the course of a year, a simplistic explanation of observed patterns should be viewed cautiously. The interaction between canopy gap fraction and GPP, independent of temperature, incident light, nutrients and grazing should be explored with manipulative experiments. The use of shade cloth, opaque plastic and streams with an alternating pattern of riparian cover (e.g. repeating sequences of open and forested reaches) shows promise in exploring the consequences of canopy modification for in-

steam metabolic processes (e.g. Grimm and Fisher 1984, Bunn *et al.* 1997, Davies and Bunn 1999).

The influence, if any, of deciduous and evergreen riparian vegetation on GPP in Australian streams remains unclear. Pidgeon (1978) showed significantly lower rates of GPP in a willow lined reach compared to eucalypt woodland lined and open reaches. The eucalypt woodland had a very open canopy with no understorey suggesting a highly degraded condition. This resulted in a canopy gap fraction of 85% compared to 26% at the willow lined reach. Pidgeon attributed observed differences in GPP to differences in PAR. In the present study the native riparian forest at WoCrN was dense and contained understorey species, with a canopy gap fraction of 7-20 %, and mean GPP was little different to that at SpCrW. However, results for both studies are inconclusive, with low statistical power and lack of replication. The scant trend toward different seasonal patterns and greater GPP in native lined reaches in some months but lower rates in others, should be seen only as a guide to further experimentation.

Riparian canopy cover explained 44-68% of the variation in GPP at a stream reach and cobble habitat level in the Mary River, Queensland (Davies and Bunn 1999). They suggested that the 'natural' metabolic processes of undisturbed forest streams are characterised by diatom dominated periphyton with low levels of GPP. When the riparian forest is disturbed the stream autotrophic community becomes dominated by high light requiring filamentous algae and macrophytes. The latter were characteristic of Pidgeon's (1978) open and woodland sites. Both WoCrN and SpCrW appear to have canopy shading characteristics and rates of GPP that are suggestive of the 'natural' condition of forested streams in Australia proposed by Davies and Bunn (1997).

Twenty four hour community respiration (CR_{24})

Estimates of CR_{24} in the present study are similar to estimates for similar sized Australian streams (Table 6.2). The comparisons discussed in relation to GPP (Section 6.4.2) are relevant to CR_{24}, with similar rates determined by Pidgeon (1978) and Chessman (1985), lower rates by Treadwell (1995) at Tweed Spur Creek, Davies

(1993, cited by Treadwell 1995) and Davies and Bunn (1999) and greater rates by Treadwell (1995) in fourth to sixth order streams and Davies (1997) in the Pilbara.

Treadwell's (1995) CR_{24} estimate of 19.7 g O_2 m^{-2} d^{-1} for Little River is the largest for an Australian stream and very high compared to overseas studies. Only two of the 30 studies in Webster and Meyer (1997a) showed comparable rates, both from streams in arid regions of USA (Cushing 1997, Jones *et al.* 1997). Similarly only one of the 17 studies tabulated by Allan (1995) recorded equivalent respiratory rates and then only on a single occasion with a lower mean. CR_{24} estimates in the present study are one fifth to one seventh of these very high values whereas GPP estimates were at least one thirtieth of the highest rates of GPP measured in Australia and overseas. This suggests a greater conservatism in CR_{24} rates than GPP between streams and biomes.

Comparison of the two sites in the present study suggests few overall differences in CR_{24}, with the exception of high CR_{24} in autumn at WoCrN. This value is responsible for the greater mean at WoCrN, but the effect size (0.2) was low. CR_{24} appeared to be greatest in summer or early autumn and least in winter at both sites, suggesting that temperature and possibly litter inputs (Chapter 4) are important factors in the seasonal pattern of CR_{24}. Autumn at both sites, prior to the first spates, was characterised by low flow with visible accumulations of coarse organic matter in the streams from summer and autumn litterfall. There appeared to be greater accumulations at SpCrW so organic matter accumulation alone does not explain the high autumn rate of CR_{24} at WoCrN. Both Pidgeon (1978) and Chessman (1985) noted coherent seasonal patterns of CR_{24} at all sites and the maximum values occurred in spring, summer or autumn with the minimum values in late autumn or winter. It is possible the apparent seasonal trends in the present study, Pidgeon (1978) and Chessman (1985) are due to lack of replication both within and between sites. Treadwell (1985) had replicated chambers within sites and showed no significant differences between seasons and contrasting seasonal trends at four sites. It is difficult to compare the unreplicated whole reach measurements with replicate chambers within a reach. The latter explores variability within a reach at small scales,

with the size of chambers typically less than 1 m^2. Whole stream methods explore variability at a reach scale (100's m^2). It is expected that sample variance will differ as the scale of the sample unit changes. Between season comparisons by ANOVA are dependent on sample unit and thus exploration of changes in variance with scale in streams would aid in synthesising results from disparate studies. Notionally, chamber methods embed variability at scales <1 m^2 within the between reach comparisons which would be expected to increase the size of the population variance. This postulate could be explored through comparative experiments.

It is difficult to reliably distinguish between CR_{24} patterns and magnitude at WoCrN and SpCrW. Pidgeon (1978) noted lower CR_{24} at a willow lined reach relative to eucalypt woodland lined and open reaches. However the reach characteristics do not compare well with the present study (Section 6.4.2), thus the influence, if any, of a deciduous versus evergreen canopy on CR_{24} in Australian streams remains unclear.

Net daily metabolism (NDM) and ratio of GPP to CR_{24} (P/R)

These two parameters have been used to distinguish the energetic basis of stream ecosystems (Odum 1956), with a positive NDM and P/R>1 indicating an autotrophic system and a negative NDM and P/R<1 indicating a heterotrophic system. Fisher and Likens (1973) modified this conceptual model to include energy flow characteristics arguing that the P/R ratio yields no understanding of organic matter transported across ecosystem boundaries. They proposed an expanded model incorporating import and export of organic matter. Allan (1995) suggested that a P/R>0.5 represents a situation where more than half the metabolic energy is derived autochthonously and that this may be a better indicator of autotrophy in a system. Further discussion of metabolic processes in relation to organic matter processing takes place in Chapter 8.

In the present study NDM was always negative and P/R ratios always <0.5, suggesting the stream reaches studied are heterotrophic and energetically dependent on allochthonous inputs. There were no significant differences between WoCrN and SpCrW in mean rates of NDM or P/R, but effect size and power were both low, giving an inconclusive result. Seasonal trends in NDM appear to mimic those of CR_{24} and similar corollaries apply, including lack of replication and the disproportionate

influence of the single autumn WoCrN value. There are no distinct seasonal trends in P/R at either site, though the spring and summer high ratios suggest autotrophy is relatively more important at these times even though absolute rates of GPP were not necessarily maximal at the time.

The similarity in mean P/R ratios appears to be further evidence that there are few differences in the annual metabolic processes in the two reaches studied. However, an impression is gained that the metabolic signal was, on occasion, stronger at WoCrN and an intensified study in which metabolic processes are the sole focus may reveal seasonally important differences that may be attributable to the phenology of the riparian vegetation.

In Chapter 8 community metabolism data will contribute to the development of partial organic matter budgets. Discussion of stream metabolism index and recycling and turnover rates of organic matter will refer to some of the subjects raised in this chapter. The next chapter (Chapter 7) presents organic matter standing crop data, the last of the components required for compiling the partial stream organic matter budgets in Chapter 8.

ORGANIC MATTER STANDING CROP

Introduction

In the previous chapters organic matter inputs to the study catchment were considered. Once delivered to the stream, organic matter will remain in transport or be physically trapped (Lamberti and Gregory 1996). Trapping of coarse particulate organic matter may be an essential pre-requisite for microbial colonisation (Lamberti and Gregory 1996) which precedes mechanical and invertebrate processing through fragmentation and consumption (Webster and Benfield 1986). The resulting dynamic between input, retention and processing ('output') represents organic matter storage (Lamberti and Gregory 1996).

In streams, accumulated organic matter is an important storage of energy and nutrients and influences physical characteristics including channel stability, fauna habitat and retention capacity (Jones 1997, Harmon *et al.* 1986, Bilby and Bisson 1998). LWD has often been considered separately because of its role in channel form, sediment and organic matter deposition and retention, and in-stream and riparian habitat (Harmon *et al.* 1986, Bilby and Bisson 1998, Marsh *et al.* 1999). Brooks (1999b) states that LWD may have been one of the dominant geomorphic controls on the structure and function of many streams and rivers in Australia This geomorphological role has been dramatically altered by the removal, intentional or otherwise, of LWD (Bilby and Bisson 1998, Brooks 1999a).

In this chapter, the standing crop of organic matter will be considered for experimental reaches within the study catchment. Storage and retention characteristics will be discussed with respect to riparian vegetation type (willows and native). Organic matter accumulated within and on the surface of the stream substrate is termed benthic organic matter (BOM). This can be further divided into fine benthic organic matter (FBOM >0.45 μm to 1mm Wallace and Grubaugh 1996) and coarse benthic organic matter (CBOM > 1mm Lamberti and Gregory 1996). CBOM may include large woody debris (LWD, frequently defined as wood >2.5 cm diameter or >10cm diameter Harmon *et al.* 1986, Bilby and Bisson 1998) and debris dams.

Methods

Benthic organic matter (BOM)

Five random locations along the channel midline were selected at each site. Working upstream to avoid disturbance to other sample locations, a 33 x 33 cm (0.1 m^2) quadrat was placed on the stream bed. If large organic matter particles such as bark, large leaves or branches were only partially within the quadrat they were picked up at their intersection with the quadrat and snapped or torn at that point and the appropriate portion retained. The substrate was then thoroughly disturbed with a gloved hand to a maximum depth of 0.1 m with a 250 μm net held at the downstream edge of the quadrat to capture disturbed sediment and organic matter. Each quadrat was re-sampled twice with the same collection effort to improve collection efficiency (ie. a total of three sample events for each quadrat).

Willow root mats at SpCrW formed a coherent massive structure sometimes the full width of the stream. This could not be disturbed by hand and required excavation with a pruning saw. Complete excavation yielded a 0.33 x 0.33 x 0.1 m (0.01 m^3) sample. During excavation contamination was minimised by using a sharp and thinly bladed saw that cut on the pull stroke which allowed accurate cutting. Once the square of root mat (0.33 x 0.33 m) was cut to a depth of approximately 0.15 m it was possible to lift it intact from the substrate. The base of the resulting rectangular prism was then trimmed to yield a final sample of the desired dimensions (0.01 m^3). Where trapped cobbles made complete excavation difficult three to five sample cores of known volume were taken and results converted to a final volume of 0.01 m^3 less the volume of all stones trapped in the upper 0.1 m of the root mat. Stones were removed from the sample location after cores had been taken and their volume determined by displacement of water.

Samples were dried at 105°C to constant weight, ashed at 500°C for 5 hours and reweighed. Ash free dry weight (AFDW) was used as a measure of organic matter content. Substrate type (bedrock, cobble, pebble, sand, silt/mud, debris dam or willow root mat) was recorded at approximately one metre intervals along each 100 m reach.

Observations of depth to bedrock were made at all sample locations, variously using depth of hand disturbance, a small metal rod and auguring to gauge the depth.

Statistical analysis was by one way analysis of variance with Duncan's *post hoc* testing (Statistica, StatSoft Inc). Homogeneity of variance within data was tested with Levene's test and where the test was significant ($\alpha = 0.05$) data were log transformed and homogeneity re-tested. ANOVA was performed on log transformed data if assumptions were met.

SaCrM willow to native transect

At SaCrM site a section of the stream was lined by *Salix fragilis* on one bank and native vegetation (*Eucalyptus viminalis, Acacia melanoxylon* and *Callistemon sieberi*) on the opposite bank. A reach of approximately 20 m length was selected with no meanders or evidence of active erosion of either bank. Five transects at two metre intervals perpendicular to stream flow were marked on the banks. BOM was sampled along the transects at four locations; the willow lined bank, willow lined channel edge, native bank and native lined channel edge.

Samples were collected with cylindrical soil cores 4.7 cm diameter and 5 cm depth. These were pressed or hammered into the sediment until flush with the surface and then excavated. Ends were trimmed flush with the cylinder surface yielding a sediment core of known volume and known surface area. Oven dry weight (105°C overnight), bulk density (g oven dry weight per cm^3), AFDW (following combustion at 500°C for 2 hours) and loss on ignition (% of sample dry weight lost during combustion) was determined for each sample. AFDW was expressed on an area basis (g AFDW m^{-2}).

Statistical analysis (ANOVA) was as described in Section 7.2.1.

Large woody debris (LWD) and debris dams

LWD and debris dams were surveyed along a one km reach of Spring Creek upstream and including SpCrW, a one km reach of Sailors Creek upstream of the Spring Creek - Sailors Creek junction and 100 m reach of Wombat Creek at WoCrN. Distance was

estimated by pacing and checked against geographic features recognisable on 1:25000 topographic maps. The location of all pieces of LWD larger than approximately 10 cm diameter and 1 m length were recorded. The pieces were described as either within the channel ('within'), bridging the channel ('bridging') or one end in the channel and the other resting on a bank ('partial'). These categories are similar to those used by Bilby and Bisson (1998).

Debris dams were always associated with LWD and were grouped under four categories: 'bankside' when caught against the bank by a piece of LWD; 'small' when containing one or two pieces of LWD and small amounts of other organic matter; 'medium' when containing several major pieces of LWD with much trapped organic matter and sometimes sediment forming a loosely packed structure often with no upstream pool or downstream scour; and 'large' when as for medium but up to 5 major pieces of LWD and a densely packed structure containing sediment and forming an upstream pool and usually a step, riffle or scour downstream.

In an attempt to quantify relative standing crops of LWD and debris dams an arbitrary value was assigned to the LWD and debris dam categories (Table 7.1) and the score summed for each surveyed reach.

	Category	Score
LWD	bridging	0
	partial	1
	within	2
Debris dams	bankside	0.25
	small	1
	medium	2
	large	3

Table 7.1. Relative scores assigned arbitrarily to categories of LWD and debris dams used during surveys of 100 m and 1000 m stream reaches within the study catchment.

Results

Benthic organic matter (BOM)

There were significant differences in the quantity of BOM at the six study sites ($F_{5,23}$ = 5.30, P = 0.002, ANOVA on log10 transformed data, Figure 7.1). Duncan's *post hoc* testing grouped willow sites (WoCrW and SpCrW) and the native and mixed sites (WoCrN, SaCrN1, SaCrN2 and SaCrM) separately. The average quantity of BOM at native sites was approximately 500g AFDW m^{-2}, at the mixed site approximately 570g AFDW m^{-2} and at willow sites approximately 5080g AFDW m^{-2}.

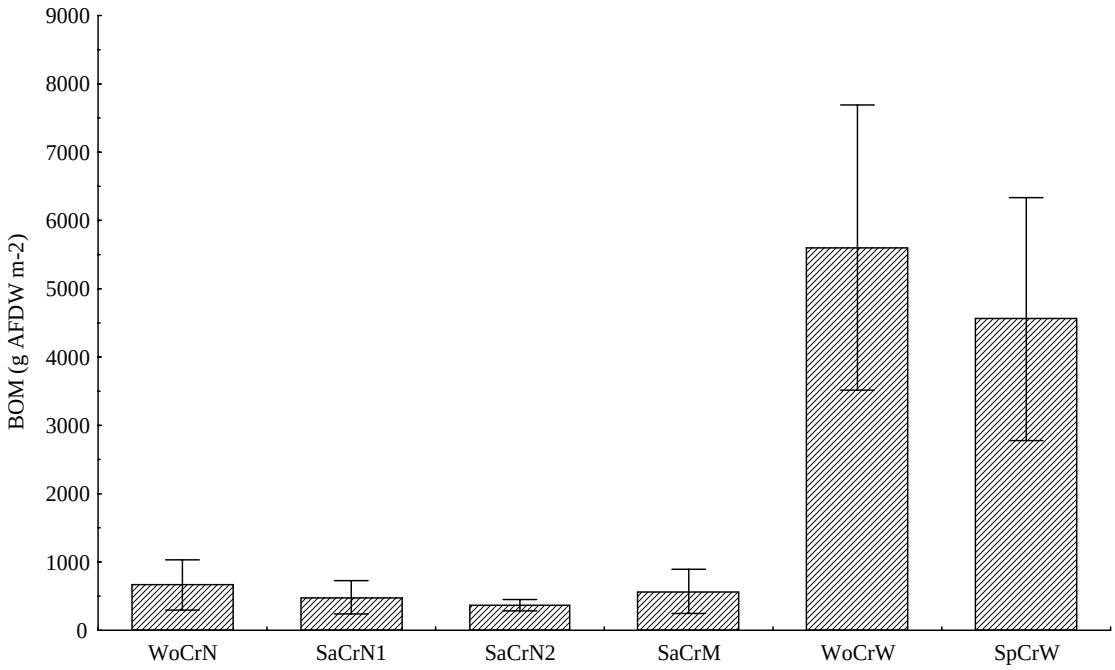


Figure 7.1. Benthic organic matter (>250 μm, g AFDW m^{-2}) sampled from the uppermost 10 cm of substrate at reaches with contrasting riparian vegetation. Three reaches were lined with native vegetation (WoCrN, SaCrN1 and SaCrN2), two with willows (WoCrW and SpCrW) and one with mixed vegetation (SaCrM). Values represent means and error bars standard errors of five randomly located 0.1 m^2 quadrats along the channel midline.

Substrate type and depth

Increased BOM at willow sites was associated with substrate characteristics. At the willow sites the dominant substrate recorded was willow root mat (76% of records at

SpCrW) or silt and sand (77% of records at WoCrW). At native and mixed sites bedrock and cobbles dominated (>75% of records at WoCrN, SaCrN1, SaCrN2 and SaCrM). WoCrN was of slightly different character in having a wide range of particle sizes present at the majority of locations. Whilst bedrock and cobbles were present in 83% of records they were frequently associated with gravel, sand and silt. In addition approximately 7% of records were 'organic debris'. Organic debris was not recorded at the other native sites or the mixed site but accumulations of organic debris were noted in 26% of records at WoCrW and at SpCrW large accumulations of debris, recorded as debris dams, constituted 6% of the substrate records. Figure 7.2 shows SaCrN and SpCrW sites in autumn 1997 during very dry conditions. The contrast between bedrock-cobble and willow root mat substrates is visible.

7.2a 7.2b

Figure 7.2. Substrate detail at SpCrW (7.2a) and SaCrN1 (7.2b). Litter traps approximately one metre across provide scale.

Substrate depth to bedrock was associated with substrate type. At sites where bedrock and cobbles dominated (SaCrN1, SaCrN2 and SaCrM) average substrate depth to

bedrock was approximately 5 cm. At WoCrN depth to bedrock was slightly ambiguous due to mass wasting of underlying shale. However, *in situ* weathered clays were readily distinguished from coarser alluvial sediment upon excavation and mean depth to bedrock was approximately 10 cm. At the willow sites auguring was needed to establish depth to bedrock with mean depth to bedrock approximately 35 cm at both sites.

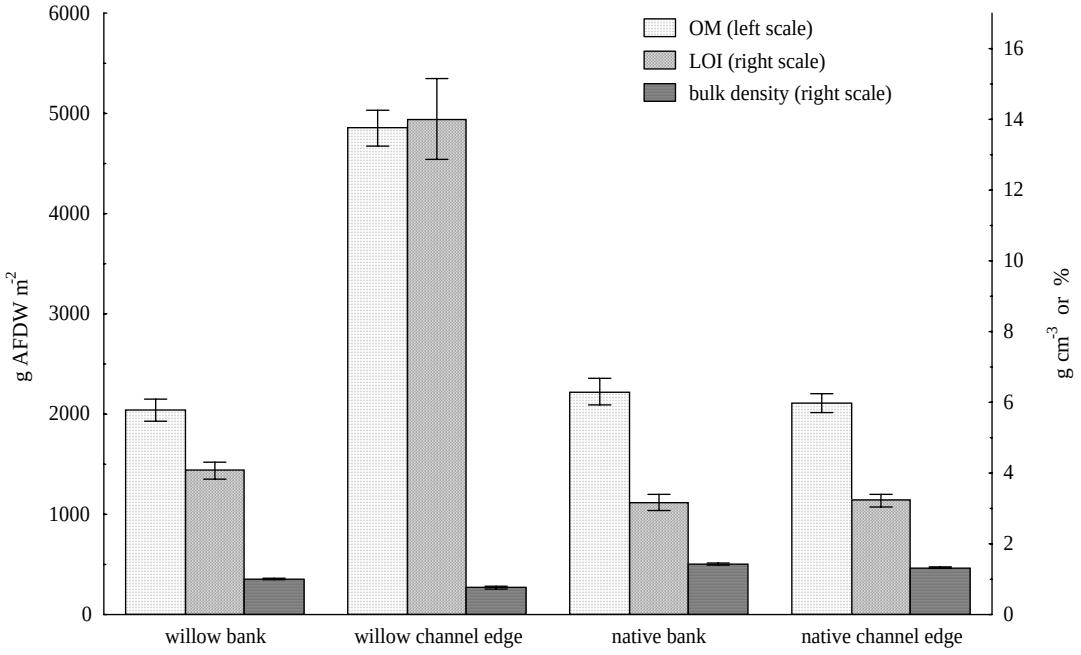


Figure 7.3. Organic matter (OM, g AFDW m^{-2}, left hand scale), loss on ignition (LOI, %, right hand scale) and bulk density (g cm^{-3}, right hand scale) for four locations across transects from a willow lined bank to an opposite native lined bank at SaCrM. Values are means and error bars standard errors for five transects 2 m apart.

SaCrM site transect

BOM ($F_{3,16}$ = 21.08, P <0.001), LOI ($F_{3,16}$ = 18.77, P < 0.001, log transformed data) and bulk density ($F_{3,16}$ = 22.28, P < 0.001) showed significant differences between sample locations across the transects (Figure 7.3). Duncan's *post hoc* testing showed willow channel edge BOM and LOI to be significantly larger and bulk density to be significantly lower than other locations. In addition bulk density of the willow bank was significantly lower than that of the native channel edge and native bank locations.

Large woody debris (LWD) and debris dams

Surveying of LWD and debris dams showed that debris dams were associated with all pieces of LWD in Spring Creek but individual pieces of LWD dominated Sailors and Wombat Creeks (Figure 7.4). Spring Creek debris dams consisted of one or more major pieces of LWD with associated branches, twigs, leaves, sediment and other debris. Debris dams were up to 1 m high, 2 m wide (ie. their axis parallel to stream flow) and spanned the full channel (2+ m). Debris dams often extended beyond the bankfull channel and appeared capable of trapping flood debris above channel banks.

Wombat and Sailors Creeks were characterised by individual pieces of LWD. These were either a fallen tree trunk or, much less commonly, a large limb. The pieces often bridged the channel or were braced on one bank with the other end in the channel. They occasionally had associated debris in Wombat Creek but rarely in Sailors Creek. The contrasting nature of LWD and debris dams is illustrated in Figure 7.2, with bridging LWD visible at Sailors Creek and within channel LWD and associated debris in Spring Creek.

LWD relative standing crop was 130 km^{-1} in Wombat Creek, 46 km^{-1} in Spring Creek and 7 km^{-1} in Sailors Creek. Debris relative standing crop was 40 km^{-1} in Wombat Creek, 44 km^{-1} in Spring Creek and 4 km^{-1} in Sailors Creek.

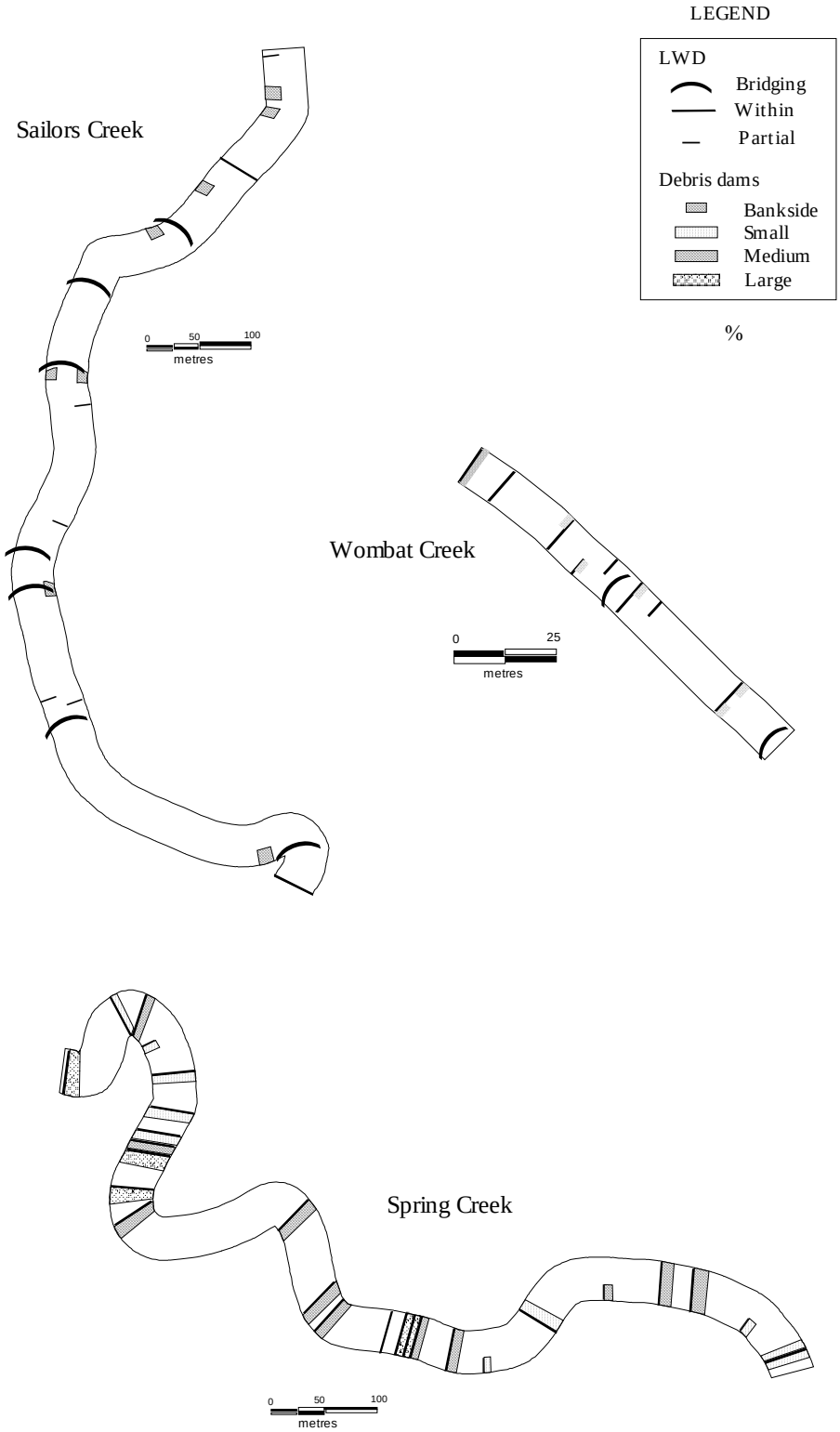


Figure 7.4. Reach survey diagrams showing the distribution of LWD and debris dams along reaches lined with native vegetation (Sailors and Wombat Creeks) and willows (Spring Creek). Stream widths exagerated up to 10 times and banks drawn parallel for clarity. Note scale is different for Wombat Creek. Reach base maps from VicMap 1:25 000 digital maps.

Discussion

Benthic organic matter (BOM)

There are few data for BOM in Australian streams (Lake 1995, Table 7.2). Pidgeon (1978) sampled organic matter in Commissioners Waters, NSW. His data were not directly comparable to the present study as methods differed. However, he estimated total BOM of 48 to 94g AFDW m^{-2} in three pools lined with either willows, pasture or eucalypt woodland. The largest single component at all sites was periphytic algae which was estimated from GPP and production to biomass (P/B) ratios. Pidgeon (1978) acknowledges that an error in GPP or the P/B ratio would have led to an equivalent error in standing crop. In addition the sampling depth was 5 cm, compared to 10 cm in the present study. Even assuming that doubling sampling depth at Commissioners Waters would double BOM the values so obtained would only be 2-38% of those in the present study.

Pearson *et al.*'s (1989) study of a tropical rainforest stream also estimated relatively low BOM in a stream riffle. They acknowledge that BOM in pools would most likely be greater but provide no additional data. The remaining Australian studies have all been conducted in central Victoria, primarily in streams lined with native forest. The results for the native forest lined reaches in the present study are very similar to the range of values for Victorian streams in Table 7.2. The estimates for the willow lined reaches in the present study are three times greater than the maximum BOM previously recorded in Australia (Reed 1989, Boulton 1988)

Jones (1997) analysed BOM from 31 stream organic budgets for the USA and other countries presented in Webster and Meyer (1997a). Non-wood BOM ranged from 3 – 5929 g AFDW m^{-2} with a median of 280g AFDW m^{-2}. The high value is equivalent to that of willow lined streams in the present study. It was determined in a steep, 1^{st} order stream in a coniferous forest on the north coast of the USA. The next highest value (2772 g AFDW m^{-2}) was from a 6^{th} order coastal river in deciduous and coniferous forest on the south eastern coast of the USA. There was a significant correlation between total BOM (including wood) and both litterfall and rainfall and a significant correlation between CPOM and wood standing crop. Wetter locations with

high plant productivity were associated with large organic matter standing crop, especially when LWD was abundant as it acts as an effective organic retention device (Jones 1997). Precipitation, litterfall and LWD were similar at native and willow sites in the present study and these parameters cannot explain the order of magnitude differences in BOM standing crop.

The contrast between reaches increases if differences in substrate depth are considered. Willow lined reaches had an average substrate depth of 35 cm and native reaches of 7 cm. On average all BOM at native sites was sampled as substrate depth was less than sampling depth. At willow sites auger spoil was visually indistinguishable throughout the channel substrate profile and coherent willow root mat was present to bedrock, indicating that organic matter content of the substrate remained high throughout the 35 cm of substrate. BOM at willow sites may be up to 3.5 times that estimated from only the top 10 cm of substrate, in which case BOM at willow sites would be approximately 35 times greater than at native sites.

The willow root mat was the key feature distinguishing native and willow lined reaches. BOM standing crop values were high when willow root mat was the substrate sampled. Visually a sizeable portion of the AFDW was the willow root mat itself. In spring new root growth was pink and was obvious against the darker older roots and could be seen growing across the wetted channel. Objects within the channel were incorporated into the root mat. A single seasons growth covered 100 mm of a piece of metal inserted into the substrate to at SpCrW to facilitate flow measurement (Section 6.2.4). Mineral sediment, bedrock, LWD, debris dams and stone walls built during the gold rush were all observed undergoing similar root mat encroachment. At SpCrW and WoCrW sites willow root mat had expanded across the full width of the wetted channel except at pools deeper than approximately 50 cm. At SaCrM site the willow root mat was fully across the wetted channel only where large willows were on opposite banks and the channel not deeper than approximately 50 cm. For this reason willow root mat and entrapped debris dams were frequently sampled in willow lined reaches whereas mineral substrates were sampled in those lined with native or mixed vegetation.

Author	Location	Method	Canopy, reach characteristics	BOM
This study	WoCrN	substrate disturbance to 10 cm,	native forest	664 g AFDW m^{-2}
	SaCrN1	0.25 mm net	native forest	483
	SaCrN2		native forest	367
	SaCrM		mixed forest	570
	WoCrW		willow	5603
	SpCrW		willow	4556
Pidgeon 1978	Commissioners Waters, NSW	0-5 cm benthos plus macrophyte	Willow, pool	48 g AFDW m^{-2}
		standing crop plus periphyton	pasture, pool	69
		from P/B ratios	eucalypt woodland, pool	94
Barmuta 1988 [a]	Acheron River, VIC	-	-	80 - 360 g DW m^{-2}
Boulton 1988 [a]	Werribee River, VIC	-	native forest, temporary reaches	up to 1500 g DW m^{-2}
Reed 1989 [a]	Central Victorian streams	-	native forest	360 - 1340 g DW m^{-2}
			open/pasture	260 - 1000
Pearson *et al.* 1989	Yuccabine Creek, QLD	substrate disturbance, 0.4 mm net	tropical rainforest, riffle	38 - 184 g DW m^{-2}
Treadwell 1995	Tweed Spur Creek	Suber sampler to 10 cm depth,	native forest	123 - 206 g AFDW m^{-2}
	Keppel Creek	0.3 mm net	native forest	108 - 146
	Little River		mixed forest	50 - 83
	Acheron River		mixed forest	163 - 177

Table 7.1 Benthic organic matter (BOM) from Australian streams and rivers. Estimates are presented as either g AFDW m^{-2} or g DW m^{-2}. [a] Cited in Lake 1995.

This observation was tested at a location within SaCrM site where willows were growing along one bank and native vegetation on the other. Organic matter content and LOI (related measures although the former was areal and the latter gravimetric) were greatest within the willow root mat at the willow channel edge (Figure 7.3). Where mineral sediments dominated the substrate (willow bank, native bank and native channel edge) organic matter content and LOI were significantly less.

The lower bulk density of willow channel edge samples compared to the native bank or channel edge could also be explained by the presence of willow root mat. Organic matter generally decreases bulk density (Handreck and Black 1991) and the net-like web of willow roots is likely to have increased porosity. The willow bank was a flood - mediated levee overlying willow terrestrial root mat. The sediments at this location appeared coarser than those of the opposite (native) bank where in situ weathered clays and colluvium were the dominant origin of sediment. Bulk density is typically lower in coarser sediments (Charman and Murphy 1991) when other influences such as organic matter content and biopores are similar.

LWD and debris dams.

The relative loading score of LWD was greater at Wombat Creek and Spring Creek than in the lower reaches of Sailors Creek. However, the LWD relative scoring system did not distinguish between an individual piece of LWD and an accumulation of more than one piece of LWD in a debris dam. The relative standing crop of debris dams was greater at Spring Creek than at Wombat Creek.

The volume of LWD within Wombat Creek, Sailors Creek and Spring Creek has since been estimated by Heath Cameron (pers. comm., University of Ballarat, November 2000). He quantified the total volume of all pieces of LWD with a diameter of 0.1 m or greater at five randomly selected 60 m reaches within willow and native riparian vegetation lined one kilometre reaches. These one kilometre reaches were all 4^{th} order and included four of the sites in the present study, WoCrN, WoCrW, SaCrN1 and SpCrW. There was a significant difference between the volume of LWD within the willow (averaged 0.007 m^3 m^{-2}) and native vegetation lined reaches (averaged 0.015

m^3 m^{-2}). The large accumulations of debris observed at the willow sites did not contain a large number of pieces greater than 0.1 m in diameter.

The dynamic between LWD recruitment, density and durability, stream power and aggradation may explain the difference between reaches. Sniderman (1998) showed willows in Spring Creek to be even aged and between 20 - 50 years old. Few trees were single stemmed, most having 3 – 5 outward leaning main trunks ≥0.5m in diameter. Willow timber has low strength (Bootle 1983) and stems had often snapped off and either resprouted or contributed to in-stream LWD. The resulting morphology of the riparian vegetation was distinct with a large number of near horizontal trunks that must be climbed over or under when walking along the stream banks (Figures 3.7b and 7.2).

Once in the stream channel, flow can move LWD if stream power is great enough. Stream power is directly proportional to discharge and gradient. ($\Omega = \gamma Qs$, where Ω is stream power, γ is specific weight of water, Q is discharge and s is slope, Knighton 1984). Stream power at mean annual discharge is similar at Wombat and Spring Creeks. Willow wood is less dense than eucalypt or wattle wood (air dry densities of 350-450, 750 and 640 kg m^{-3} for willow, *Eucalyptus viminalis* and *Acacia melanoxylon* respectively, Bootle 1983) and a lower stream power would be required to move an equivalent sized piece of LWD. Reorganisation of willow wood at Spring Creek would be expected to be greater. A further confounding factor is aggradation, for this has the effect of trapping LWD in position until it decomposes or is physically broken into smaller pieces. Both the accumulation of sediment within debris dams and willow root mat entrapment operate in trapping LWD along Spring Creek and the willow lined reaches of Wombat Creek. In addition, the entrapment and subsequent burial of LWD under sediment or willow root mat meant that pieces of LWD present within the channel were not visible and so did not contribute to the relative loading score, yet may remain functionally significant.

Recruitment at native vegetation lined reaches of Wombat and Sailors Creeks occurred through tree fall from forest self-thinning and wattle senescence. The high rainfall and deep soils of the Wombat Creek valley have led to a tall dense canopy

and eucalypts out competed in the race to fill the canopy have died and fallen as whole trees. Eucalypt timber is generally high strength and observations suggested fallen tree trunks generally remained intact as they fell with only limbs and crowns breaking. *Acacia dealbata* was a relatively common understorey species and is generally short lived (approximately 20 years). It again mainly fell as whole trees with main trunks intact. Wombat Creek is narrow (mean width ~2 m) and bridging LWD was formed when trunks fell across rather than within banks. Limb and crown material were observed to have fallen into the channel following a heavy snow fall in winter 1998, but main trunks appeared to dominate in stream and terrestrial LWD standing crop.

The combination of dense timber and a channel with relatively low stream power allowed LWD to be retained in the native vegetation lined reaches of Wombat Creek and Sailors Creek. There was evidence of sediment accumulation behind logs but relatively few debris dams. Sediment accumulations had part buried logs but all LWD was still visible. The relative loading score for Wombat Creek would thus reflect a larger effective recruitment period than that of Spring Creek. Eucalypt timber is more durable than willow (Bootle 1983) and the larger LWD loading score and higher volumes recorded by Cameron (pers. comm., University of Ballarat, 2000) at Wombat Creek may reflect longer turnover times for individual pieces of LWD.

Recruitment of LWD at the surveyed reach of lower Sailors Creek was similar to Wombat Creek with tree fall common and whole trunks dominating in-stream and terrestrial LWD standing crop. Lower Sailors Creek was wider (up to 7 m) than Wombat Creek and upper Sailors Creek (both approximately 2m) This increased the likelihood that at least some part of a tree would fall into the channel. Stream power in Sailors Creek at mean annual discharge was nearly three times that of Wombat Creek. The wider channel meant stream power per unit width did not differ but also meant fewer obstructions were present to trap LWD. Transport during spates could explain the relatively low LWD standing crop in lower Sailors Creek compared to Wombat Creek.

The combination of stream power, channel width and lack of obstructions such as LWD would also explain the low relative standing crop of debris dams. No debris dams had accumulated appreciable sediment or influenced channel morphology along lower Sailors Creek whereas debris dams and LWD in Spring Creek and Wombat Creek, were responsible for pool formation and channel steps (LWD forcing, Montgomery and Buffington 1998).

The future status of LWD and debris dams is linked to riparian succession. Due to the two-fold impact of the 1850's gold rush and early 20th century logging and grazing few old trees exist in the native riparian forests of the study catchment. As the forest continues to mature the potential size of LWD will increase. Larger pieces are better able to resist stream power and may initiate debris accumulation in higher order reaches. At present key large pieces of LWD initiating debris dams were observed only in lower order reaches ($\leq 4^{th}$). Channel roughness and number of obstructions can also increase the likelihood of LWD retention and can be increased by riparian vegetation encroachment. This was exemplified in the study catchment by willow stem collapse and in-stream colonisation but was also observed in *Callistemon* and *Leptospermum* thickets such as those illustrated in Figure 3.6b.

Willow encroachment ('choking') has been seen as a detrimental process in Australian streams (Ladson *et al.* 1997) yet in the light of the changes wrought by riparian clearing and de-snagging (Brooks 1999a, Marsh *et al.* 1999) the process of encroachment may also be seen as restorative. Cohen (1999) illustrated channel recovery mechanisms in a Victorian forested stream that were directly attributed to within channel vegetation and LWD. In Cohen's study colonising and later successional species were native but the recovery mechanisms, particularly sediment storage in the channel network and increased channel roughness were very similar to the willow mediated processes observed in the present study.

Sniderman's (1998) observations of succession in willow forests of Spring Creek implies that near monoculture willow forests are a temporary state if later successional species are available for recruitment. The fate of willow root mat

dependent BOM and LWD standing crop following succession in willow forests has not been studied. Similarly, there are no data on consequences of willow clearing on BOM and LWD. The telescoping ecosystem model, a conceptual model developed by Fisher *et al.* (1998) linking material cycling and disturbance, predicts that disturbance decreases total nutrient storage and increases processing length of materials. Current willow management strategies focusing on poisoning and removal (disturbance) will, logically, lead to release of stored organic matter and increase processing lengths. Management directed toward tolerating, directing or accelerating succession will lead to the reverse. Over the past decade residents near Spring Creek have explored the use of shade tolerant native and exotic trees planted under undisturbed willow canopies to direct forest succession toward multiple function forests. Species with most potential have included *Acacia melanoxylon, Pomaderris aspera., Bedfordia arborescens, Nothofagus cunninghamii, Araucaria bidwillii, Juglans nigra, Populus alba, Sequoia sempervirens, Fagus sylvatica* (D. Holmgren, pers. comm., Hepburn Springs, 1999).

The standing crop data in this chapter are incorporated into partial organic matter budgets prepared in the next chapter. That chapter's discussion further considers organic matter processing efficiency and retention characteristics of willow and native vegetation lined reaches.

PARTIAL ORGANIC MATTER BUDGETS

Introduction

Three partial organic matter budgets are presented. One each for a 'typical' reach lined by willows and by native vegetation. These can be considered reach or point budgets (Webster and Meyer 1997c). A third budget is presented for the 98 km^2 Wombat–Sailors–Spring Creek catchment (watershed budget, Webster and Meyer 1997c). It has been acknowledged earlier that there is variation within both willow and native riparian vegetation in terms of structure and floristics (Section 3.5 and 4.4.2). The concept of a 'typical' reach may thus be called into question. However, it has been used because some organic matter inputs and storages quantified in earlier chapters have shown a higher degree of similarity within these vegetation types than between them. For example, seasonality of litterfall (Chapter 4) and BOM storage (Chapter 7) were more similar within willow sites and native sites than between them. At other times significant differences were not detected either within or between riparian vegetation types and hence the values used in the construction of the point budgets were little different. Finally, the two riparian vegetation types differ in canopy phenology (deciduous versus evergreen) and in root mat morphology, which were both important influences on the organic matter parameters quantified in the present study. It was thus assumed that despite the floristic and structural diversity within the riparian vegetation types the categories 'willow' and 'native' represent two groupings with more in common than between.

The assumptions and techniques applied to data from earlier chapters are outlined in the following methods section. The budgets are then used to compare the sources, storage and processing of organic matter within the reaches and the catchment as a whole.

Methods

Reach budgets

Mean annual data (converted to g AFDM) for litter input, groundwater input and GPP input for native and willow reaches were obtained from Chapters 4–6. Groundwater input was assumed to be evenly distributed across the catchment despite the point discharges from mineral and QBA springs.

GPP was converted from units of g O_2 m^{-2} y^{-1} using:

$$g\ C = g\ O_2 * 1/PQ * 12/32$$

where PQ (photosynthetic quotient) = 1.2, atomic weight of carbon = 12 and molecular weight of oxygen = 32 (Bott 1996). One g C was assumed to equal two g AFDM (Webster and Meyer 1997b).

Community respiration was converted from g O_2 m^{-2} y^{-1} using:

$$g\ C = g\ O_2 * RQ * 12/32$$

where RQ (respiratory quotient) = 0.85 (Bott 1996).

GPP and community respiration at all forested sites was assumed to the same. The more open canopy and wider valley of the lower reaches of Sailors Creek may have increased light levels and GPP in some seasons. This may have been compensated for by the temporary nature of much of Sailors Creek which would have reduced GPP to zero in late summer–early autumn and restricted respiratory activity to the scant hyporheous. Willow canopy condition was similar at all willow lined locations and willows lined spring fed permanently flowing reaches. GPP and CR_{24} for willow and native vegetation lined reaches was assumed to be the mean of values from WoCrN and SpCrW.

Mean areal mass of benthic organic matter (BOM) standing crop was obtained from Chapter 7. Mass of LWD and debris dams were not independently quantified. However, debris dams and channel debris accumulations in pools and against obstructions were included in BOM determinations if incorporated in randomly

located sample quadrats. Relative loading scores for LWD and debris dams are included in both budgets but were not used for further calculations. The values for Spring Creek were used for all willow reaches and the mean of the Sailors and Wombat Creek values were used for native reaches.

Catchment budget

Stream riparian vegetation for the whole catchment was mapped by ground survey and interpretation of aerial photography. All riparian vegetation was identified as either native, willow, mixed or agricultural and the stream length associated with each category determined. Means weighted by riparian vegetation type for each of the budget parameters were produced. It was assumed that mean stream width was uniform across all vegetation types. As native vegetation dominated the lowest (1st) and highest (5th) stream orders and the other communities generally occurred in the middle reaches (2nd-4th) this assumption was reasonable.

Groundwater inputs were uniform across all vegetation types. Native and willow means were determined from the reach budgets (Figures 8.1 and 8.2). Litterfall for mixed vegetation was obtained from Chapter 4 and BOM standing crop from data in Chapter 7. GPP and CR_{24} for all forested reaches (willow, native and mixed vegetation lined reaches) were assumed to be the mean of WoCrN and SpCrW sites.

Ground survey in the agricultural riparian communities was difficult with a large number of small holdings and absentee landlords. Aerial photograph interpretation and general observation was sufficient to map these areas. Approximately 50% of the agricultural riparian vegetation was cleared (treeless) and the rest treed. The trees were mostly native (*Eucalyptus* and *Acacia melanoxylon* dominated) but included some deciduous exotic species. Litterfall for agricultural reaches was assumed to be 50% of the native litterfall. Benfield (1997) found no correlation between CBOM and litterfall but a significant correlation between CBOM and wood. In the present study wood relative standing crop was high at WoCrN but low in the lower reaches of Sailors Creek, yet BOM was not significantly different. Thus for agricultural reaches,

despite lower litterfall and presumed low levels of wood, BOM was assumed to be the same as the mean for native reaches.

Additional light and potential nutrient enrichment suggests that GPP would be greater at agricultural reaches than at native or willow reaches (see Section 6.4.2). Davies (1997), using data from Western Australian streams, suggested moderate catchment impacts such as small scale catchment clearing and nutrient input could be expected to increase GPP two to ten times. Davies and Bunn (1999) presented a predictive model for a Queensland stream where GPP increased by an order of magnitude if forest was completely cleared. This was applied in the present study and it was further assumed that GPP remains unchanged where the riparian zone was treed. Thus GPP for the agricultural zone was estimated as:

$$GPP_{ag} = 0.5 * (10 * GPP_{native}) + 0.5 * (GPP_{native})$$

Davies and Bunn (1999) suggested that CR_{24} was less responsive than GPP to catchment clearing resulting in an increase in the P/R ratio. Davies (1997) suggested a P/R ratio of 1-1.5 in moderately degraded catchments. An assumption that P/R = 1.5 was used for agricultural reaches (i.e. $CR_{24} = 1.5*GPP_{ag}$).

LWD and debris dam relative loading scores were not estimated for the catchment as they were a comparative tool and would be misused if algebraically manipulated to give an absolute score for the catchment.

Net primary productivity (NPP), heterotrophic respiration (R_H) and autotrophic respiration (R_A)

In open water methods used for community metabolism measurements no separation of autotrophic and heterotrophic respiration is possible. Assumptions used in the present study were that NPP = GPP - R_A, R_A = 50% GPP and $R_H = CR_{24} - R_A$ (Webster and Meyer 1997b)

Autochthonous versus allochthonous inputs

The relative importance of autochthonous and allochthonous contributions to reach productivity were assessed using P/R ratio (GPP/CR_{24}) and the ratio of autochthonous to total inputs (NPP / NPP + litterfall + groundwater DOM).

Ecosystem efficiency

Four indices relating to material cycling were obtained from Webster and Meyer (1997c) and Sinsabaugh (1997). Ecosystem efficiency (CR_{24} / total inputs x 100) was calculated for the catchment but not for the reach budgets. For a reach, comparison of community respiration to total inputs is inappropriate because organic matter being consumed in respiration may be imported from upstream. If import from upstream is included as an input then, depending on reach length, much material may be exported without being consumed. Measurements of efficiency will be low in short reaches and will increase with reach length as the probability of entrapment increases. Consequently, ecosystem efficiency is as dependent on reach and stream length as on ecosystem processes

Stream metabolism index.

Fisher (1977) proposed stream metabolism index (SMI) as a measure based on the ratio of organic matter consumed by respiration to organic matter inputs in excess of those that would maintain a constant rate of organic matter in transport across the system (Elwood *et al.* 1983). Webster and Meyer (1997c) suggested SMI could be considered the ratio of observed respiration to respiration needed to prevent accumulation of organic matter. Cummins *et al.* (1983) defined SMI for a stream reach as:

$$SMI = CR_{24} \pm \Delta S / GPP + L + G + T + M - Q_f M$$

where ΔS is change in storage, L is litter (including lateral movement), G is groundwater DOM, T is tributary inputs, M is inputs transported into the reach at the upstream site and Q_f is the ratio of the discharge at the downstream site to discharge at the upstream site.

Whilst 100m reaches were the experimental unit used to gather data presented in earlier chapters in the present study, the budgets presented for a typical willow and typical native site are essentially point budgets. For a point budget $Q_f = 1$ and tributaries are absent. Groundwater inputs can be considered to enter the stream at discrete point sources (spring eyes, etc.) within a catchment and thus, like tributaries, are absent from a point budget (Webster and Meyer 1997c). Alternatively, groundwater inputs can be seen as entering uniformly along a stream margin, (as for GPP and litter), especially if base flow times average groundwater DOM content is the basis of calculation. For the present study, uniform groundwater input was calculated and was included in the calculation of SMI for the two point budgets. As benthic organic matter standing crop data in the present study is for a single year, no change in storage was calculated and ΔS was assumed to be zero (Webster and Meyer 1997c). Consequently, SMI was calculated as:

$$SMI = CR_{24} / GPP + L + G$$

For a catchment budget, groundwater is the only upstream input so that $G = M$, $Q_f = 1$ and there are no tributaries. If it is again assumed that ΔS is zero SMI can be calculated as:

$$SMI = CR_{24} / GPP + L$$

Specific respiration

Specific respiration (or BOM recycling rate, Sinsabaugh 1997) relates respiration to benthic organic carbon standing crop such that:

Specific respiration (y^{-1}) = respiration ($g\ m^{-2}\ y^{-1}$) / BOM ($g\ m^2$).

Sinsabaugh (1997) used heterotrophic respiration as the numerator. Community respiration (CR_{24}) includes autotrophic respiration which is not derived from consumption of benthic organic matter. Specific respiration can be considered as the rate that available food (BOM) is consumed relative to the mass of potential food present. This figure allows inter system comparison of heterotrophic efficiency that is independent of organic matter retention efficiency.

BOM turnover time

BOM turnover time can be calculated based on biological processing or both processing and transport. The former is the inverse of specific respiration and gives an estimate of the time taken for a unit mass of BOM to be converted to CO_2 through biological processing (heterotrophic consumption). Turnover based on processing (T_P) was calculated from:

$$T_P\ (y) = BOM / R_{H.}$$

Turnover based on processing and transport (T_{PT}) can be calculated based on organic matter inputs or outputs (Webster and Meyer 1997c). Inputs are necessarily used in the present study but Webster and Meyer (1997c) suggested inputs were preferable because of the dynamic nature of outputs. T_{PT} was calculated as:

$$T_{PT}\ (y) = BOM / \text{total inputs}.$$

Results

Native versus willow reach budgets.

Total inputs for native and willow reach budgets were similar (Table 8.1). BOM standing crop was an order of magnitude greater for the willow budget and represents the most obvious difference between the two budgets (Table 8.1). LWD relative standing crop was larger for the native budget but was underestimated at willow sites. LWD generally occurred as individual pieces at native sites and as multiple pieces in debris dams at willow sites (Chapter 7).

Indices relating to the relative importance of autochthonous and allochthonous inputs (P/R and NPP/total inputs, Table 8.3) were 0.2 for native and willow budgets.

Stream metabolism index (SMI) was approximately one in both budgets. Indices dependent on BOM were noticeably different between native and willow budgets, reflecting the large difference in BOM standing crop. Specific respiration was an order of magnitude lower and BOM turnover an order of magnitude greater for the willow budget.

	Native	%	Willow	%	Catchment	%
Inputs						
GPP (g AFDW m^{-2} y^{-1})	153	21	153	20	256	32
Litterfall (g AFDW m^{-2} y^{-1})	442	60	477	62	416	51
Groundwater (g AFDW m^{-2} y^{-1})	137	19	137	18	137	17
Total inputs (g AFDW m^{-2} y^{-1})	732	100	767	100	809	100
Standing crops						
BOM (g AFDW m^{-2})	500		5080		849	
LWD (km^{-1})[a]	69		46		-	
Debris dams (km^{-1})[a]	22		44		-	
Outputs						
CR_{24} (g AFDW m^{-2} y^{-1})	741		741		819	
R_A (g AFDW m^{-2} y^{-1})	77		77		128	
R_H (g AFDW m^{-2} y^{-1})	664		664		691	

Table 8.1. Stream partial organic matter budgets for 'typical' locations with native and willow riparian vegetation and for the Wombat-Sailors-Spring Creek catchment as a whole. Data are from the present study (Chapters 4-7) with assumptions described in Section 8.2. GPP = gross primary production, BOM = benthic organic matter, LWD = large woody debris, CR_{24} = community respiration, R_A = autotrophic respiration, R_H = heterotrophic respiration. [a] relative loading score km^{-1} (Chapter 7).

Riparian type	Stream length (km)	% Stream length	GPP (g AFDW m^{-2} y^{-1})	Litter (g AFDW m^{-2} y^{-1})	BOM (g AFDW m^{-2})	CR_{24} (g AFDW m^{-2} y^{-1})
Native	79	68	153	442	500	741
Willow	8	7	153	477	5080	741
Mixed	12	10	153	486	570	741
Agricultural	17	15	842	221	500	1263
Total	116	100				

Table 8.2. Data used to calculate Wombat-Sailors-Spring Creek catchment organic matter budget components (Table 8.1). Stream length and percent of total stream length lined by four riparian vegetation types and values for organic matter budget components for each riparian vegetation type. All native and willow values and litter and BOM values for mixed riparian sites derived directly from experimental chapters (Chapters 4, 6 and 7). GPP and CR_{24} at mixed sites and all values for agricultural sites based on assumptions explained in Section 8.2.

	Native	Willow	Catchment
P/R	0.2	0.2	0.31
NDM	-1.61	-1.61	-1.54
NPP/total inputs	0.11	0.10	0.16
Ecosystem efficiency (%)	-	-	101
SMI	1.01	0.97	1.22
Specific respiration (y^{-1})	1.33	0.13	0.81
T_P (y)	0.75	7.65	1.23
T_{PT} (y)	0.68	6.26	1.05

Table 8.3. Indices of stream condition derived from partial organic matter budgets (Table 8.1) for 'typical' locations with native and willow riparian vegetation and for the Wombat-Sailors-Spring Creek catchment as a whole. P/R = ratio of GPP to CR_{24}, NDM = net daily metabolism (GPP - CR_{24}), NPP = net primary production, SMI = stream metabolism index, T_P = BOM turnover based on processing, T_{PT} = BOM turnover based on processing and transport.

Catchment budget

Organic matter inputs (Table 8.1) to the Wombat – Sailors –Spring Creek catchment budget were dominated by litter fall (51% of total inputs) with autochthonous (GPP 32%) and groundwater (17%) inputs contributing proportionally less. Total inputs were larger than those for native and willow reach budgets, the result of an approximately 1.7-fold increase in estimated GPP.

Estimated respiratory output was dominated by heterotrophic respiration (84% of CR_{24}) and was larger than those for the native and willow reach budgets. Standing crop was similar in magnitude to total inputs and respiratory output and was between the standing crop estimates for the native and willow reach budgets.

P/R (0.31) and NPP/total inputs (0.16) ratios indicate allochthonous inputs dominated the catchment. Ecosystem efficiency was ~100% and SMI was >1 (1.22) suggesting that community respiration consumed organic matter at a rate not matched by measured organic matter inputs. Specific respiration and BOM turnover (T_P and T_{PT}) were approximately unity, suggesting that respiration consumed BOM standing crop in approximately one year and combined processing and transport consumed BOM

standing crop in a similar period.

Discussion

Inter-catchment comparison

Comparison of organic matter budgets and derived indices between catchments has more to do with what has and what has not been included and the accuracy of parameter estimates than any biological or physical differences (Webster and Meyer 1997c). Inter catchment comparison will need improved data collection techniques, larger data sets and consideration of the variability of parameters across the stream network within a catchment (Webster and Meyer 1997c, Cummins *et al.* 1983). For example SMI >1 and ecosystem efficiency ~100% reflect unmeasured inputs rather than material cycling processes (Webster and Meyer 1997c). In the present study, unmeasured inputs include wood and willow roots. Specific respiration and BOM turnover time (processing basis) spanned five orders of magnitude for 21 organic matter budgets reviewed by Sinsabaugh (1997) with results for the present study within the upper quartile for specific respiration and lowest quartile for BOM turnover. BOM turnover time based on processing and transport (BOM/total inputs) varied over two orders of magnitude in Webster and Meyer's (1997c) review of 30 organic matter budgets with results for this budget similar to the median value. Turnover was longest in those streams with large amounts of wood reflecting the lack of input data for wood and slow recycling and transport rates for LWD.

This contrasts with the usefulness of short-term partial budgets when comparing selected processes between diverse stream ecosystems (Cummins *et al.* 1983). As a result the following discussion will only briefly address inter-catchment comparisons and will focus on riparian vegetation types within the catchment.

The only published organic matter budget for an Australian stream (Treadwell *et al.* 1997) estimated 75% of total organic matter inputs were from groundwater DOM. This contrasts with the results for the present study in which inputs were dominated by litterfall. Reasons for the differences between catchments in estimates of

groundwater DOM contribution are discussed in Chapter 5. In Treadwell *et al.*'s study DOM transport was 2.8×10^6 g y^{-1} which could be accounted for by a DOM input of 170 g m^{-2} y^{-1} evenly across the total streambed area (16 450 m^2). This was only 3% of the estimated input of 5507 g m^{-2} y^{-1} implying that 97% of the DOM input was retained or processed. Heterotrophic respiration and non-wood storage could not explain this unaccounted for DOM. The unusually high groundwater input was reflected in a low ratio of NPP to total inputs, low ecosystem efficiency and low BOM turnover based on transport and processing.

Treadwell *et al.*'s (1997) results were used earlier in the present study to postulate that if large groundwater DOM inputs were a general feature of Australian streams these could mask changes to litterfall dynamics that may result from willow colonisation (Chapter 2). This was not supported in the present study, as groundwater DOM did not dominate organic matter inputs to the streams.

Intra-catchment comparison

Organic matter standing crop clearly distinguished native and willow reach budgets. The order of magnitude greater BOM standing crop at willow sites had an influence on specific respiration and BOM turnover time of similar magnitude. Specific respiration of 0.13 y^{-1} for the willow reach budget suggests 13% of the standing crop was processed (i.e. converted to CO_2) by heterotrophs each year. The turnover time of BOM on the basis of processing was 7.7 years. Transport can also remove BOM standing crop and thus turnover time on the basis of both processing and transport should have been, and was, less (6.3 years). For the native reach budget specific respiration was 1.33 y^{-1} and turnover times 0.75 y (processing) and 0.68 y (processing and transport). Short turnover time for organic matter in native reaches is consistent with the observation that little organic matter appeared to accumulate from year to year.

In addition, SMI, arguably a robust measure of reach scale processes (Fisher 1977, Webster and Meyer 1997c), was approximately one for both native and willow reach

budgets. This suggests that the observed respiration rates were equal to that required to prevent organic matter loading (Webster and Meyer 1997c) yet organic matter (BOM, LWD and debris dams) had clearly accumulated at all sites. Two explanations are proposed.

The first is that the years of data collection were unusual, such that estimates of CR_{24} were greater than normal and/or GPP or litter fall were less than normal. Data were collected in low rainfall years, which could have reduced riparian vegetation vigour and decreased litterfall. Cummins *et al.* (1983) suggested wet year inputs from litterfall and lateral movement exceeded dry year inputs by 5% and 65% respectively for an Oregon forest watershed. A 5% increase in litterfall would decrease SMI by only 3–5% for all budgets. As lateral movement was insignificant even large increases would have little effect on SMI. How a dry year affects GPP and CR_{24} is uncertain but the magnitude of the effect would need to be large and probably GPP lowered and CR_{24} elevated to yield substantial change to SMI.

The second explanation is that organic matter inputs may have been underestimated. Wood inputs were not quantified. Whilst biological processing of wood can be slow and restricted to the outermost few millimetres (O'Connor 1992, Cummins *et al.* 1983) the large mass of wood potentially available may contribute to measured CR_{24} but was not a measured input. Cummins *et al.* (1983) showed breakdown and leaching of wood may contribute substantially to DOM and FPOM loading in streams but the input of wood is ignored in the calculation of SMI and other material cycling indices. Input of organic matter via willow roots was another important unquantified input. For the willow reach budget, root mats contributed to estimates of BOM and presumably to CR_{24}. When alive, roots respire, thus directly contributing to measured respiration and when dead become potential foods for heterotrophs. Root mats in the stream channel are derived from GPP of the riparian forest canopy. Stream organic matter budgets should include the biomass of roots added to the stream channel prior to loss by transport or processing (i.e. the gross root productivity). Net root productivity over the 50 years since willow colonisation is reflected in the 35 cm

deep willow root mat dominated substrate at SpCrW. This and the willow bank–native bank transect (Chapter 7), showed that the presence of willow root mats explained the large differences in BOM observed at native and willow sites.

Recycling rate of BOM (specific respiration) was lower for the willow budget than the native budget and hence BOM turnover time based on processing was larger. This suggests the standing crop of organic matter at willow sites was processed slower than an equivalent standing crop at native sites. At willow sites the lower layers of sediment appeared anaerobic, with black roots and a sulphurous odour. Minshall *et al.* (1983) suggested BOM in anaerobic layers would decay slowly and suppress recycling rates for the location. The shallow coarse sediments of native sites are unlikely to ever be anaerobic and recycling rates may thus be relatively high. Smith and Lake (1993) compared surface and buried (to 10 cm) leaf packs in a Central Victorian stream and found no difference in weight loss with time and greater grazing intensity in the buried leaf packs.

Willow leaves are consumed rapidly in Australian waters (Pidgeon and Cairns 1981, Yeates 1994, Schulze and Walker 1997) and the low durability of willow wood (Bootle 1983) implies it is decomposed readily. The breakdown of willow roots has not been investigated in Australia. If unpalatable and slow to decompose this will also suppress recycling rates at willow sites.

BOM turnover time based on both processing and transport was larger for the willow budget than for the native budget. This may be explained by low biological cycling of willow root mats and/or by physical resistance to transport. The latter is the reason for extensive willow plantings for erosion control (Ladson *et al.* 1997) and is consistent with the observation that the willow root mat along Spring Creek had accumulated to a depth of approximately 35 cm (Chapter 7).

Two dominant influences are apparent in the material cycling indices for the catchment. GPP at cleared sites and BOM at willow sites were an order of magnitude

above other sites and exert a strong influence on the catchment budget. GPP at cleared sites was modelled from Davies and Bunn (1999) and the assumptions need to be confirmed for Victorian waters. If valid then clearing of ~10% of the riparian vegetation in the study catchment had a measurable influence on the relative importance of autochthonous and allochthonous inputs. P/R ratio for the catchment was 0.31 compared to 0.2 for forested reaches and NPP / total inputs was 0.16 compared to 0.11 and 0.10 for forested reaches. Minshall *et al.* (1983) noted that small streams with low P/R ratios are extremely important processing locations along a stream continuum. As P/R ratios increase streams become producers rather than processors. Clearing of very small proportions of the riparian zone in a catchment may have a large influence on the ratio of production to processing.

Material cycling indices for the catchment reflected the large organic matter standing crop at willow lined reaches. The catchment budget has a mean BOM standing crop 1.7× that of the native reach budget, yet willow lined reaches constitute only 7% of the catchment. SMI recycling rates and turnover times for the catchment budget are between those of the willow and native reach budgets, reflecting greater R_H and intermediate BOM standing crop.

The next chapter draws conclusions from this and earlier experimental chapters, from both an ecological and management perspective.

CONCLUSIONS

Ecological conclusions

The organic matter budget approach used in the present study was successful in providing new data and perspectives on ecosystem processes influenced by an invasive riparian species. Organic matter inputs, organic matter storages and metabolic processes were estimated for reaches lined by willows, by native vegetation and by a community that included both of these elements. These data were up-scaled to the level of the catchment and used to estimate indices of material cycling. This approach has identified characteristics of organic matter cycling that were most strongly influenced by riparian vegetation type in the study catchment. This was a novel use of organic matter budgets. It avoided many of the inherent problems of comparing stream organic matter dynamics across large temporal and spatial scales (Cummins *et al.* 1983, Section 2.4). As data were gathered simultaneously at reaches within a single mid-sized catchment (~100 km^2) there were relatively uniform hydrological and geomorphological influences. Additionally, all reaches had a similar disturbance history, most notably the intensive working of the channel and riparian sediments for gold during the 1850's gold rushes. The native sites were not 'reference sites' devoid of extensive anthropogenic disturbance. The influence of riparian vegetation type could thus be separated from the influence of disturbance (*sensu* Fisher *et al.* 1997). The absolute values for parameters within these budgets must be considered within the particular climatic conditions and time scale in which they were gathered (Chapter 3). It was the relative values between reaches that were the emphasis of the present study and the results were successful in exploring postulates concerning willow impacts on stream ecology.

Organic matter storage, litter quality (as indicated by ash content), and seasonal light penetration of the canopy differed significantly between reaches lined by willows and native vegetation. Total litterfall did not differ significantly and there was less distinction in seasonality of litterfall than expected given the obvious phenological difference between the deciduous and evergreen canopies. Metabolic processes (GPP and CR_{24}) were quantified for two reaches and were valuable in studying average metabolic processes within forested reaches (85% of streams in the catchment).

Assumptions generated from the literature were used to estimate community metabolism within the cleared reaches (Section 8.2.2). The estimates suggested that there were large metabolic consequences of riparian canopy cover removal from a relatively small proportion of streams in the study catchment. These conclusions are explored in an ecological context and from a management perspective in the following sections.

Annual total litterfall was not significantly different at sites lined with willows, native or mixed riparian vegetation. Litter accession into the study catchment was within the range recorded for forested streams in south eastern Australia and overseas. Phenology exerted an influence on, but was not a universal indicator of the seasonal pattern of litter accession into the experimental reaches. Litterfall was less seasonal than expected from northern hemisphere studies (e.g. Campbell 1993). When *Eucalyptus* - *Acacia* forests had an understorey of *Leptospermum* and *Callistemon* (SaCrN2) seasonal patterns were identical to a similar canopy with an understorey of willow (SaCrM). Where riparian forest communities along low order streams are degraded by understorey clearing, willow colonisation would potentially have the same impact on seasonality of litterfall as *Leptospermum* and *Callistemon* colonisation. From a biodiversity perspective, the latter would be regarded as preferable but in terms of gross organic matter accession (mass and timing of organic matter delivered) the two communities may be less distinguishable.

A key point that is not indicated by amounts and timing of litterfall is the 'quality' of the organic matter delivered to the stream biota. In agriculture, food quality includes consideration of palatability, digestibility and nutrient content (Mackenzie 1970). Willow leaves have been shown to be palatable to biota in Australian and New Zealand streams (Pidgeon and Cairns 1981, Yeates 1994, Lester *et al.* 1994a, Schulze and Walker 1997). The ash content analysis in the present study (Section 4.4.1) is evidence that willow leaves, flowers and bark may have nutrient contents significantly different from the equivalent litter components of *Eucalyptus*, *Acacia*, *Leptospermum* and *Callistemon* species growing in riparian zones.

Catchment scale biogeochemical budgets have been well integrated with organic matter budgets in the Hubbard Brook Experimental Forest (Gosz *et al.* 1972, Likens 2001). Studies of such longevity and complexity have not been attempted in Australia (Harris 2001). Given the extent of willow invasion and subsequent modification of the riparian zone, it could be expected that large scale changes in stream biogeochemical cycles may have been initiated. Elements such as nitrogen and phosphorous have particular human importance through their contributing role in noxious algal blooms (Senate Standing Committee on Environment, Recreation and the Arts 1993). Other organisms have particular requirements for elements (for example crustaceans and calcium, diatoms and silica) and their role in ecosystem function may be modified if cycling of these elements is disrupted.

Summer dominance of litterfall in *Eucalyptus* - *Acacia* riparian forests has been widely acknowledged (e.g. Lake 1995, Campbell *et al.* 1992). This coincides with periods of low flow in south eastern Australian streams. In temporary streams summer and autumn litter accumulated on the stream bed or in remaining pools and was flushed downstream as flow returned which led to an autumn pulse of organic matter (Boulton and Suter 1986). The present study showed a similar pattern would be expected at willow sites. The net result of an autumn-spring dominated rainfall pattern and summer-autumn dominated litterfall appears to be an autumn pulse of organic matter in the surface stream at willow and native vegetation lined reaches. This is particularly true where the latter have a well developed structure and are floristically complex. As floristic complexity increased litterfall dynamics appeared to become more similar, regardless of the proportion of exotic or native species in the riparian community.

These observations are important for interpretation of biological assessments that have been conducted in waterbodies under willow and native canopies. Where invertebrates are used to indicate stream condition such as in the AUSRIVAS models (AUSRIVAS n.d.), it is useful to separate influences from riparian vegetation type (a proximate cause) from those of broader catchment dysfunction (distant causes). Pidgeon (1978) found significant differences in both material cycling and invertebrate communities in an unreplicated study of a willow-lined, eucalypt-lined and open

reach in NSW. The experimental design confounds decisive conclusions, but given that the reaches were in close proximity on the same stream, catchment conditions would be similar and differences should be attributable to reach differences only. The lowest invertebrate biomass was associated with the reach with maximum litter accession. Later work, using leaf packs, showed that willow litter was not less palatable than native litter (Pidgeon and Cairns 1981) and that macroinvertebrate biomass and community structure could not be explained by lack of available food. More litter, despite it's palatability, did not equate with more macroinvertebrates. Macroinvertebrate biomass was greatest in the reaches that were autotrophic (GPP exceeded CR_{24}). Effects of canopy removal on macroinvertebrate abundance and community structure have been identified in other studies where open and forested reaches have been compared (Behmer and Hawkins 1986, Bird and Kaushik 1992, Reed *et al.* 1994). Productivity of palatable epiphyton was considered a possible contributing factor in these studies. The present study suggests that reaches that have a dense canopy cover, whether willow or native, functioned similarly in terms of gross organic matter accession and in-stream community metabolism. In cleared reaches these parameters were estimated to be very different and, consequently, macroinvertebrate communities would be expected to be very different.

Invertebrate biomass along the littoral zones of large rivers in the Murray-Darling Basin have not been shown to be different at sites lined by willows and native vegetation (Besley 1992, Hardwick *et al.* 1995, Schulze and Walker 1997). Despite this each of these studies suggested that a dense willow canopy potentially influenced periphyton productivity. The diatom community colonising Weeping Willow and Red Gum litter was shown to be different (Schulze and Walker 1997). If periphyton composition is affected by leaf characteristics then this impact is potentially 'exported' beyond the site of litterfall. It is possible to envisage a relatively heterogeneous drift of exotic and native litter along the River Murray with littoral invertebrates responding to large scale upstream conditions as much as immediate riparian conditions. The extent of willow invasion along the River Murray may mean local differences between reaches lined by willows and native vegetation have been masked by larger scale ecosystem changes (Schulze and Walker 1997). The present study suggests that such effects could be a result of litter quality rather than quantity

or timing. The consequences of changes in litter quality for biogeochemical cycling and biota have not been well investigated in Australia and this is a notable knowledge gap.

Litterfall dominated organic matter inputs at all sites and contributed 50% of total inputs to the surface stream component of the catchment. This is consistent with theoretical and empirical data from forested low order streams in the northern hemisphere (Vannote *et al.* 1980, Webster and Meyer 1997a), but differs from the only published estimate for an Australian stream (Treadwell *et al* 1997, Section 8.4.1). GPP contributed <25% of total inputs at forested sites. The estimate for the catchment as a whole increased to 35%. This increase resulted from a predicted ten fold increase in GPP at cleared sites, despite the fact they occupied <10% of total stream length. This implies that profound influences on community metabolism at a catchment scale can result from small scale riparian clearing. Investigation of the impact of clearing on community metabolism at a reach and catchment scale has begun in Australia (Davies and Bunn 1999) but has not been placed in the context of catchment scale organic matter dynamics. The present study indicates the potential importance of this knowledge gap. Data could be obtained from interspersed cleared and forested reaches to determine the net impact of clearing on metabolic processes. If conducted in the study catchment or one similar, this would explore the validity of the assumptions used earlier in this study (Section 8.2.2).

Groundwater DOM contributed 16% of the total inputs to the surface stream component of the catchment. This differs markedly from one estimate for the nearby and biophysically similar Keppel Creek (75%, Treadwell *et al.* 1997). By critically appraising the methods used to estimate groundwater DOM it was found that the differences were due to the methods used rather than fundamental catchment differences. Fortuitously, data were available to recalculate DOM contribution using common assumptions (Section 5.4.4). Depending on the set of assumptions invoked, DOM contribution varied by over an order of magnitude. Despite this point having been made in general terms for organic matter budget parameters (Webster and Meyer 1999c), it has not been illustrated quantitatively for DOM contribution. Where comparisons are attempted between studies, between catchments or between

continents then differences in methods completely confound biophysical interpretations. This is a major failing of most published organic matter budgets (Cummins *et al.* 1983, Webster and Meyer 1999c). Interesting questions on catchment function, stream ecosystem productivity and aquatic biogeochemistry require sound knowledge of DOM input and transformation (Findlay and Sinsabaugh 1999). Common methods, probably as part of long term ecological research projects, must be used if significant advances are to be made through inter-system comparisons (Cummins *et al.* 1983, Webster and Meyer 1999c). However, it must be again stated that the approach taken in the present study utilised identical and synchronous methods to investigate riparian vegetation types within a system, and was not dependent on direct comparison with other studies or other catchments.

Retention of sediment and organic matter in willow root mats and debris dams resulted in aggradation at willow sites. Little aggradation was observed at native sites with low LWD loading. Where LWD loading was relatively high, native sites retained more sediment but still less than one third that of willow sites. Entrapment within willow root mats was considered more important than LWD in explaining differences in aggradation between the sites.

Bedrock defined channels in the study catchment were assumed to be induced by gold rush scouring. Decades of aggradation at willow sites was reflected in the depth of sediment to bedrock (average of 35 cm compared to seven cm at native sites). This may have led to a marked change in the morphology of the channel and adjacent small floodplains (generally 10 – 50 m wide). Firstly, aggradation would lead to an increase in the volume of the hyporheus, especially in bedrock defined channels. Secondly, aggradation, debris dams and willow establishment within the channel appear to have increased the frequency of channel migration across the narrow and discontinuous floodplains. Finally, the floodplains themselves appeared flatter and composed of coarser sediments at willow sites than those at native sites. Hypothetically, all these observations could be attributed to aggradation with both the floodplains and channels storing accumulated sediment and organic matter (Harmon *et al.* 1986).

One consequence of these geomorphological differences is that the flow path of groundwater to surface streams would be different at willow and native sites. A longer flow path through riparian and hyporheic sediments allows greater opportunity for DOM constituents to be transformed (Fiebig 1995, Fraser and Williams 1998). The present study shows that short flow paths (< 10 m) through riparian sediments can increase DOM content of groundwater three-fold. The net consequence of this at a catchment scale was a 50% increase in the groundwater DOM contribution to the surface streams compared to the stream corridor ecosystem as a whole. A reasonable postulate is that such floodplain-derived DOM will be more labile than that derived from more distant sources as a result of the reduced time it is exposed to biodegradation within groundwater. Hence, large scale changes in catchment DOM characteristics may be mediated by modification of riparian and hyporheic sediments within a few metres of stream channels. These changes may then be 'exported' downstream through modification of DOM load or composition in the water column. Biodegradability of DOM could be determined for groundwaters sampled along flow paths from geological material distant from the stream, at the boundary of the stream corridor ecosystem and within riparian, parafluvial and hyporheic sediments. This would be necessary to determine the biological significance of observed changes in groundwater DOM that have resulted from willow mediated aggradation. DOM constitutes a large proportion of the organic carbon in aquatic systems and is recognised as affecting food webs, secondary production and nutrient retention and release (Findlay and Sinsabaugh 1999). DOM is the basis of the 'microbial loop' in stream ecosystems (Boulton and Brock 1999) and can have significant effects on the quality of and ability to treat drinking water (Nelson *et al.* 1993). With respect to DOM a focus on processes within riparian sediments, and how these are affected by willow invasion, would be important in catchments similar to that of the present study.

Restoration implies a knowledge of pre-impact conditions, which are not directly available for the study catchment. Studies on other south eastern Australian catchments suggest many Australian surface streams prior to European settlement were narrow and shallow, highly retentive, frequently discontinuous and their form controlled largely by LWD (Brieley 1998, Brooks 1999a, 1999b, Cohen 1999,

Nanson and Doyle 1999, Vincin 1999). If this was the condition of the study catchment prior to the gold rush then restoration of retention capacity at native sites appears slow. Recruitment and turnover of LWD frequently requires forest succession and senescence for maximum recruitment (Trotter 1990, Webster *et al.* 1990). Native forests in the present study appear to be still maturing from nineteenth and early twentieth century disturbance. It was observed that riparian trees were rarely more than 50 cm in diameter at breast height and had mature but not senescing canopies. The volume of LWD within fifth-order native vegetation lined channels appeared too small to exert a strong influence on retention capacity. Occasional large tree falls had delivered LWD that was resistant to stream power and had accumulated debris dams and sediment. Smaller branch fall and small tree debris are more easily moved by spates until large keystone pieces of LWD are present. In smaller order reaches (eg WoCrN) branches and small trees appeared to resist spates and had accumulated sediment and debris dams on their upstream sides. The importance of LWD in controlling gradient drops, step-pool sequences and other aspects of longitudinal channel profile decreases with increasing stream order (Harmon *et al.* 1986).

Retention capacity at willow sites was relatively high and reaches were characterised by relatively high loadings of dimensionally complex debris dams. Willow riparian vegetation was undergoing seral change with conspicuous evidence of large trunk and limb collapse (Sniderman 1998). The rapid incorporation of debris within willow root mats secured even relatively small pieces (<20 cm in diameter) against downstream transport. These delivery and securing factors are potentially offset by rapid rotting of willow wood. Native LWD accumulations were characterised by individual logs up to 40 cm in diameter. These had sometimes accumulated small amounts of sediment and organic matter on their upstream side and appeared well keyed into the bank or channel substrate. These large pieces of predominantly eucalypt wood will rot less quickly than willow wood but delivery and securing mechanisms may be very different. The dynamics, hydraulic role and geomorphological role of LWD are receiving increasing attention in Australia (eg O'Connor 1992, Gippel *et al.* 1992, Brooks 1999a, 1999b). Its role and dynamics at willow sites should not be ignored. Where willows are removed, LWD will not be delivered until newly planted native species are mature or senescent. Depending on the decomposition and fragmentation

rates of existing debris dams, stream reaches will conceivably be devoid of LWD for a period of time, which is ecologically and geomorphologically undesirable.

Management implications

The organic matter budget approach in the present study has provided a quantitative approach to evaluating the relative magnitude of organic matter impacts from willow invasion. This is the first time this has been achieved in an Australian context and can contribute to stream rehabilitation research and planning. Current planning frameworks for catchment management and steam rehabilitation emphasise goal setting (Ladson *et al.* 1999) and priority setting (Rutherfurd *et al.* 1999). The extent of willow invasion (lining an estimated 30 000 km of Victorian river frontage, Ladson *et al.* 1999) and the large scale movement of asexual propagules downstream from existing stands means catchment and regional planning strategies are particularly relevant. Priority setting requires quantitative knowledge of impacts, costs and benefits from willow invasion at both reach and catchment scales.

Willow lopping, poisoning or removal will have immediate consequences for retention capacity and community metabolism within streams. Riparian forest disturbance results in an increase in material processing length (Fisher *et al.* 1998), alteration of LWD recruitment (Webster *et al.* 1990), and a shift in autotrophic community structure and function (Davies and Bunn 1999). Estimated slow recovery of retention capacity at native sites in the present study implies that revegetation following willow removal will not maintain retention capacity in the short term. Retention capacity will probably be restored with in-stream vegetation (eg Cohen 1999) or LWD recruitment (Naiman and Décamps 1997, Brooks 1999a, 1999b). The present study showed that recruitment of LWD at native sites capable of exerting control on retention capacity equivalent to 50 year old willow sites had not occurred. This was despite 150 years since riparian deforestation and channel metamorphosis during the gold rush and 50 years since any riparian timber harvesting.

Control of reach scale community metabolism appeared independent of riparian vegetation type but removal of vegetation was predicted to dramatically alter metabolic processes. Revegetation following willow removal will restore control

when canopy closure occurs. The temporal scale to achieve this is dependent on stream width, planting density and species selection, but is unlikely to be achieved within a decade even on narrow streams. Removal of willows potentially facilitates establishment of native riparian vegetation but simultaneously decreases metabolic control and retention capacity within the surface stream. Establishing native species whilst retaining willows for the maintenance of canopy cover of the channel, allochthonous inputs and LWD delivery, may not be an insurmountable challenge. Establishing preferred forest species on the upland sides of willow fringing vegetation and under-planting of shade-tolerant, K-selected native species has been successful within the study catchment (Holmgren, D., Hepburn Springs, pers. comm. 1999). Future selective management of willows may be possible without compromising retention capacity and metabolic control. This is an attempt to direct succession from 'pioneer' willows toward a native species dominated community, without a destructive harvest phase.

The positive roles of willows have been implicitly acknowledged in management reports with suggestions that willows are preferable to no woody riparian vegetation (eg Brizga *et al*. 1998, Ladson *et al.* 1999). There is an increasing consensus on the need for care in broad-scale willow removal (Ladson *et al.* 1999, Willow Management Task Force 2000). This is balanced by declaration of most willows as weeds of national significance (National Weeds Strategy Executive Committee 2000). This places management authorities in a difficult position, especially as there is inadequate information to systematically evaluate benefits and costs of willow removal at multiple scales. A recommendation from the present study is that the positive roles of riparian vegetation, including willows, in material retention, LWD recruitment and metabolic control be recognised. Estimates for these parameters for reaches and the catchment as a whole have been attempted in the present study and these could be used to explore issues for other catchments where willow management is planned. The size of a metabolic shift or the size of a potentially released pool of sediment and OM may be able to be estimated when willow removal is proposed. However, there are no data for what constitutes a level of unacceptable change in these parameters. The discussion of determining levels of unacceptable change in the Draft Australian Water Quality Guidelines for Fresh and Marine Waters (ANZECC

1998) provides a potentially useful framework that could be applied to willow research and management. Of particular value would be MBACI (or 'beyond-BACI', Underwood 1997, ANZECC 1998) designs targeting parameters most likely to be influenced by willow removal.

Where increasing stream retention capacity is seen as restorative, willow removal may be harmful. The reverse is also true; where retention capacity is too large willow removal may be beneficial. Effective management will need to move beyond a tolerance of willows only when a clear erosion hazard would result from their removal. Quantification of unacceptable changes to a range of variables potentially under riparian vegetation control needs to occur. From the perspective of the present study, these should include litter quality, in-stream community metabolism, channel aggradation, LWD dynamics and riparian DOM sources, each of which appeared to be strongly influenced by riparian vegetation type. Changes to any of these variables could be beneficial or detrimental depending on the context. For example, willow retention capacity was considered beneficial in controlling incision in tributaries of the Goulburn River above Lake Eildon, but detrimental in reducing channel capacity downstream (Erskine *et al.* 1993). An additional multi-disciplinary challenge is thus to determine when a change in a parameter is beneficial and when detrimental. Quantifying the impacts of willows at multiple levels of ecosystem organisation and evaluating under which circumstances such changes are beneficial or harmful is a potentially fruitful combination.

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